



Economic Impact Analysis of New Jersey's Proposed 20% Renewable Portfolio Standard

Prepared for NJ Board of Public Utilities - Office of Clean Energy

December 8, 2004



CENTER FOR ENERGY,
ECONOMIC & ENVIRONMENTAL POLICY

Edward J. Bloustein School of Planning and Public Policy

THE STATE UNIVERSITY OF NEW JERSEY
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The report was prepared by:

The Center for Energy, Economic and Environmental Policy
The Edward J. Bloustein School of Planning and Public Policy
Rutgers, The State University of New Jersey

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Executive Summary

Many states, including New Jersey, have adopted a Renewable Portfolio Standard (RPS) and others are considering such measures. Although the specifics vary by state, a RPS requires that a certain percentage of the electricity sold in the state be produced from renewable resources. In April of 2003, the New Jersey Governor's Renewable Energy Task Force recommended that the existing Class 1 RPS be increased to 20% by the year 2020 from the existing level of 4% in 2008. The Board of Public Utilities (BPU) decided to conduct an economic impact analysis prior to considering such an increase.

The BPU, Office of Clean Energy engaged the Center for Energy, Economic and Environmental Policy (CEEPP) at the Bloustein School of Planning and Public Policy at Rutgers, the State University of New Jersey, to conduct an economic impact analysis of increasing the RPS to 20% in 2020 (proposed 20% RPS) compared to the existing RPS. CEEPP enlisted the participation of the Rutgers Economic Advisory Services, and formed an interdisciplinary team to perform this analysis.

This report discusses, and where possible, quantifies the incremental costs and benefits between the proposed 20% RPS and the existing RPS. Increasing the amount of electricity sold in the state that is generated by renewable technologies would increase the cost of electricity. This additional cost would, in theory, reduce the expected growth rate of the state's economy. On the other hand, reducing emissions from fossil fuel generation, primarily at plants fired by natural gas, would reduce harmful emissions and provide a benefit by avoiding the associated costs due to those emissions. Increasing the RPS would, if combined with the state's economic development policies, also attract jobs in the renewable sector of the economy. It would also provide some price pressure on the cost of natural gas, might avoid some transmission and distribution expenditures, and increase reliability for those facilities with solar photovoltaic (PV) panels.

Projecting the future over a period of 15 years is a difficult exercise and requires making assumptions regarding many key parameters that are inherently uncertain. The analysis contained in this report constructs several different scenarios regarding expected fuel prices and technological improvements. The base case assumes increasing real fuel prices and expects technological improvements that lead to cost reductions based on the renewable market assessment that is part of this project.

The costs of requiring additional renewable resources beyond the existing standard are relatively easy to quantify compared to the benefits of such a policy. There is a large degree of uncertainty within the proposed 20% RPS of quantifying benefits. Those benefits are primarily but not exclusively avoided environmental costs. Nonetheless, the wide range in available estimates of benefits should not be construed as implying that these benefits are "less real" than the costs. Clearly, policymakers must consider the associated uncertainties of both the costs and benefits in their deliberations.

Under the base case, the proposed 20% RPS compared to the existing RPS would raise electricity prices approximately 3.7% by the year 2020 but have a negligible impact on the growth

of New Jersey's economy. Under the proposed 20% RPS, the location in New Jersey of all of the manufacturing, operations, and maintenance facilities and employees needed to support the PV and wind infrastructure would add approximately 11,700 jobs and augment the economic benefits to the New Jersey economy in the year 2020. The proposed 20% RPS would cause natural gas prices to edge slightly downward for New Jersey consumers by reducing the use of this fuel in power generation. Finally, the proposed 20% RPS would increase reliability by providing electricity from PVs when the grid power is not available and may reduce expenditures on transmission and distribution (T&D) within the state.

The economic and electricity price impacts of the proposed 20% RPS, however, depend substantially on whether expected technological improvements and other factors occur that reduce the cost of PVs and wind power. For instance, if additional cost reductions do not exceed the pace of those that have historically occurred to date in PV and off shore wind technologies, the proposed 20% RPS would raise electricity prices by approximately 24% in the year 2020 and have a measurable, negative impact on the state's economy.

The proposed 20% RPS would also reduce the emission of many pollutants in the region. The marginal fuel in the region used to generate electricity is primarily natural gas, and a proposed 20% RPS avoids the emission of many major air pollutants from natural gas powered plants. Quantifying in dollars the incremental avoided cost of the proposed 20% RPS requires many assumptions, New Jersey specific data, and extensive modeling. Using estimates of externality "adders" - estimates of the additional costs due to emissions - that are found in the literature, illustrative calculations suggest that the avoided costs due to a proposed 20% RPS are in the range of several hundred million dollars in 2020.

The amount of the environmental benefits also depends on the policy interaction between the proposed 20% RPS and existing environmental policies. For instance, under the emission cap and trade programs for sulfur dioxide (SO₂) and nitrogen oxides (NO_x), these environmental externalities are internalized. Additional policy measures are needed in conjunction with the proposed 20% RPS to maximize its environmental benefits.

There are other initiatives that New Jersey policymakers should consider in order to maximize the benefits of the proposed 20% RPS, should it be adopted. These include taking measures to coordinate with other states and the federal government and to promote measures that lead to the anticipated decline in the costs of renewables.

Chapter 1: Introduction to the Economic and Environmental Assessment of a Renewable Portfolio Standard for New Jersey

I. Overview of New Jersey's Renewable Portfolio Standard

Many states, including New Jersey, have adopted Renewable Portfolio Standards (RPS) and others are considering such measures. Although the specifics vary by state, a RPS requires that a certain percentage of the electricity sold in the state be produced from renewable resources. New Jersey's RPS divides renewable resources into two classes. Class 1 includes PVs, solar thermal electric, wind, geothermal, fuel cells, landfill gas recovery and sustainable biomass. Class 2 technologies are hydroelectric and waste-to-energy. Within Class 1, there is a separate PV requirement, which requires 90 megawatts (MW) of capacity (equivalent to approximately 180,000 megawatts-hours (MWh) of PV) by the year 2008. Class 2 technologies can make up no more than 2.5% of a suppliers total RPS obligation, which is 6.5% in the year 2008. Load serving entities (LSEs) are permitted to meet their RPS commitments through facilities located within the state, within the Pennsylvania-New Jersey-Maryland (PJM) wholesale market, or from facilities that can demonstrate delivery to PJM.

The enforcement mechanism for the RPS is Renewable Energy Credits (RECs). To satisfy their specific RPS amount, LSEs must acquire a sufficient number of RECs for each type of resource requirement. They can obtain RECs by operating renewable resources or purchasing them from the market. Owners or operators of renewable projects will be issued RECs based on the production of their facilities, which they can sell in order to finance their projects. Also, LSEs that have more RECs than their specific requirement can sell them to other LSEs that cannot meet their requirements.

In April 2003, the Renewable Energy Task Force appointed by Governor McGreevey recommended i) the existing Class 1 RPS be increased from 2% to 4% in 2008, and ii) that the Class 1 RPS be increased to 20% by the year 2020. The Board of Public Utilities (BPU) acted upon the first recommendation. This increased the total Class 1 and Class 2 RPS to 6.5% in 2008. The BPU also decided to have an analysis of the economic impact of the proposed 20% RPS conducted before acting on the Task Force recommendation. The BPU engaged the Center for Energy, Economic, and Environmental Policy at the Bloustein School of Planning and Public Policy at Rutgers, the State University of New Jersey, to conduct an economic analysis of increasing the RPS to 20% in 2020 (proposed 20% RPS) compared to the existing RPS. The Center for Energy, Economic and Environmental Policy (CEEPP) enlisted the Rutgers Economic Advisory Service to form an interdisciplinary team to perform this analysis.

The purpose of this report is to conduct an economic assessment of the proposed 20% RPS, which includes identifying and, where possible, quantifying the incremental costs and benefits of the proposed 20% RPS compared to the existing RPS. The costs to the New Jersey economy of requiring additional renewable resources beyond those in the existing standard are relatively easy to quantify compared to the benefits of such a policy. Projecting costs of technologies, some of which are still maturing, some fifteen years into the future remains a difficult exercise, however. The uncertainties in quantifying the benefits of the proposed 20% RPS, particularly those that are avoided environmental costs, are large but the wide range of estimates available for the task should not be construed as implying that these benefits are "less real" than the more readily quantified

costs. Clearly, policymakers must consider the associated uncertainties of both the costs and benefits in their deliberations.

The analysis of whether to adopt the proposed 20% RPS should not be limited to a comparison of its costs and benefits. Policymakers legitimately may want to also consider a strategic perspective and think about whether existing electricity markets should be transformed and whether the proposed 20% RPS is an appropriate means to do so. This strategic view could consider the shaping and acceleration of an electric power system based on sustainable resources rather than on depleting fossil fuels as policy objective, a goal that the proposed 20% RPS would seem to advance.

II. Organization of this Report

This report is organized into four chapters. The remainder of this chapter reviews several selected studies of the impacts of proposed RPS in other states. While comparing studies across states is difficult as explained further below, it is important nonetheless to understand previous work in order to place the approach and results of this report in context.

Following this introductory chapter, Chapter 2 presents the economic assessment of adopting the proposed 20% RPS compared to the existing RPS. It investigates the statewide and renewable-sector economic impacts of the proposed 20% RPS. The analysis is based primarily upon the use of the Rutgers Economic Advisory Service Econometric Model of the New Jersey Economy (R/ECON™). This economic forecasting model has been widely used in other New Jersey-based studies to forecast key macroeconomic variables such as gross state product, employment levels, and prices. It can also be used, as in this report, as a tool to understand the economic implications of various policies. To provide detail on the economic impact of the proposed 20% RPS on the renewable resource sectors of the New Jersey economy, additional economic analyses were conducted. Chapter 2 presents the overall results and analysis first, and then proceeds to report on the more detailed assumptions and modeling issues

Chapter 3 analyzes the environmental benefits of the proposed 20% RPS. By avoiding the use of traditional fuels to generate electricity, the proposed 20% RPS provides environmental benefits by reducing emissions that result in health, environmental, and economic damage. The chapter thoroughly reviews the environmental externality literature, which can be divided into two parts. First, externality adders are presented and discussed. These adders include environmental costs associated with the use of a particular generation resource, for example natural gas, which are not internalized as part of the producer's cost structure. They are then applied to the proposed 20% RPS to illustrate the quantification of the possible range of incremental environmental benefits compared to the existing RPS. The remainder of this chapter presents a review of the "bottom-ups" literature that is used to determine these environmental adders. Similarly to Chapter 2, this chapter presents the overall results and key analysis first, and leaves the more detailed discussions to later sections.

Chapter 4 discusses the policy implications of the proposed 20% RPS. To capture fully the potential benefits of this increased standard, policymakers may have to take additional measures beyond its adoption. This chapter describes several policy initiatives for consideration and presents them in several contexts. The first context includes measures that New Jersey could adopt unilaterally to improve the effectiveness of the RPS. The second context is cooperation and coordination among states within the region. This requires New Jersey to participate in coalitions and in some cases to develop and lead coalitions to persuade the federal government and governments of other states to change specific policies both to avoid eroding the benefits of the proposed 20% RPS and to enhance its effectiveness. The chapter also highlights the importance of coordination between and among existing policies and these proposed initiatives. Chapter 4 concludes the report. Two appendices provide additional analysis and description of the economic assessment conducted in Chapter 2 and the environmental benefits of the proposed 20% RPS conducted in Chapter 3.

III. Review of Selected Renewable Portfolio Studies from Other States

Development of Renewable Portfolio Policies in Different States

Interest in renewable energy resources has been growing for many reasons beyond those of the increasing economic viability of related electricity generation technologies. One is the increasing stringency of environmental regulations covering electricity generation, starting with the Clean Air Act of 1970. Another, and one that has been heightened in the wake of the tragedy of 9/11/01, is a national security policy goal. This goal is to reduce the nation's reliance on foreign-based energy resources, particularly crude oil, although oil-fired generation is only a small percentage of the nation's total production of electricity. Of course, the reality of uncontrollable vicissitudes in oil prices and past experience with supply disruptions due to embargos also play a role in this regard. A third reason is the concern with the safe operation nuclear energy facilities after the Three Mile Island meltdown. Safe disposal of nuclear waste, which caused the costs of electricity based on nuclear fuel to skyrocket, discouraged plans for more such plants.

These rationales culminated in support for two key pieces of legislation that aided in the adoption of renewable energy: the Public Utility Regulation and Policy Act of 1978 (PURPA) and the Energy Policy Act of 1992 (EPAAct). PURPA requires electric utilities to purchase from non-utilities any renewable electricity generated below the utility's prevailing electricity costs, typically in long-term contracts. When these contracts became uneconomical, the resulting legacy for policymakers was a reluctance to use long-term contracts to achieve their policy objectives. EPAAct has a provision for production tax credits for wind power, which facilitates financing of new wind turbines.

Finally, federal dollars in the 1980s underwrote much of the early development of renewable energy technologies, with the intent of jump-starting them. That early vision and investment has made it possible for some technologies, like wind, to be economically viable now in certain areas of the country and in most areas within the foreseeable future. Even long-awaited

active-solar technologies appear to be on the verge of being economically practical nationwide.

Despite the viability of some renewable energy technologies, the market has been slow to embrace them. This may in part be because to the now deregulated electric generation market. Market participants may be cautious about long-term power purchase agreements with generators in general and the use of unproven technologies with high capital costs (wind power fits into this category), in particular. On the other hand, a guaranteed demand for power generated by a specific type of technology could, to some degree, force the hand of suppliers in favor of such technologies.

An RPS is viewed by some as just the champion needed for renewable technologies. An RPS, which is specified by a state, is designed to require that a certain percentage of electric power supplied, must derive from defined renewable resources. An RPS has become an effective and popular tool for promoting renewable energy. RPS policies tend to be structured with some flexibility with regard to the precise technology set to be used to achieve their mandates. This is important because states are not equally endowed with the same renewable resources. Maine, for example, has long successfully relied on renewable resources to generate electricity needs. Not only does it have ample hydroelectric resources, but among all states obtains the largest proportion of its electricity from biopower – non-hydro, non-wind, non-solar renewable resources – due to the strength of its forestry-related industries. Meanwhile, due to a lack of pertinent resources, New Mexico cannot possibly have much hydro-electric or biomass power generation. On the other hand, most of that state receives a lot of sun and heating-degree days, and geothermal energy is not far from the surface there. Hence, geothermal and solar power generation technologies tend to be more economical in New Mexico than they are in Maine. In Pennsylvania, waste coal qualifies under its proposed RPS.

Many states have adopted an RPS, and more will do so. Because of variations in climate and natural resource endowments, to date each state has opted to specify its own unique RPS. Some states, like Maine, started off by specifying a high RPS in part because,

prior to specifying its RPS, renewable energy use there was already well underway. Others, like California, specified an aggressive RPS (one that phases in renewables rather quickly) because of the heightened need and demand there to achieve environmental goals. Yet others, like Hawaii, did so because in the process of establishing their RPS they discovered that renewable electricity generation was more practical from an economic standpoint, even presently, than were traditional fossil-fuel technologies.

Other states currently are identifying an appropriate RPS because it is only now and with a minor subsidy that renewable energy is economically feasible. Some states intend to adopt renewables rather slowly since they prefer to wait until solar photovoltaic power becomes more practical

This report is not the first of its kind. Other states also have undertaken studies of their proposed RPSs. Iowa, for example, enacted legislation as early as 1983 in the support of renewable energy and revised it in 1991 in part due to its ethanol policies. Minnesota, adopted a similar law in 1994. And while nationwide economic impact studies of renewable energy technologies were undertaken by early as the 1980s, it seems not to have been until a 1995 study for the State of Wisconsin that any state-based RPS economic impact studies were conducted. That study and two other early ones on Arizona and Minnesota appear well summarized in the study for Hawaii by GDS Associates (2001), so they are not reviewed here. These studies enumerated the tons of various pollutants that would not be discharged and how the new technologies would affect electricity rates and monthly bills. Arizona's study also estimated the number of additional jobs and income that would occur from operating the new plants compared to conventional alternatives that otherwise would have come online.

In the following sections, seven more recent studies are summarized—those for Massachusetts, Hawaii, Maryland, New York, Vermont, Pennsylvania and Colorado. Interestingly, each study takes a different tack, even the three performed by the same study team. These studies were selected on three grounds (1) the state's geographic proximity to New Jersey, (2) the originality of the analysis, and (3)

the comprehensiveness of the study. The reports are summarized below in order of the date they were published. The focus of each summary is the RPS definition, its practical implementation in terms of technology, the nature of scenarios used in the study, and a basic summary of the findings.

Comparing different states' RPS and associated analyses is difficult to do for several reasons. First, the RPSs of states vary in the definition of what constitutes a renewable, the percentage of required renewables, and implementation schedules. Second, the cost structure of existing electricity resources and of renewables varies among states. Third, the studies analyze different RPS impacts using different methods. With these limitations in mind, there are several observations that can be made based upon the subsequent review of different RPS studies.

With the exception of that for Hawaii, the reviewed studies conclude that an RPS would increase electricity production cost and its price, although the rate increases are slight. Economic impact analyses due to these increases are not typically performed. If reductions in emissions are discussed in a study, quantifying the avoided costs in human, economic, and ecological terms is not performed. Again except for the case of Hawaii, natural gas fired generation is assumed to be the fuel-technology combination that would be displaced by an RPS. Wind power generally was found to be the dominant renewable technology alternative in meeting each RPS. This was particularly the case in the eastern coastal states and for Colorado. Solar does not tend to be significant in any of these reports because solar technologies are too expensive even through 2020. In the absence of a solar RPS requirement, solar would not be able to compete with other renewable technologies such as wind. Moreover, except for the case of Colorado, none of the state reports specify a separate solar-RPS such as New Jersey's RPS has. Perhaps one of the most common findings, again outside of those for Hawaii, was that the economic viability of the RPS relies more on the cost of natural gas than it does on the size of the RPS to be met.

Massachusetts

One of the earlier studies that analyzed a state's electric-power renewable portfolio standard was that for Massachusetts (Smith et al., 2000). According to the report, the RPS schedule in Massachusetts was to meet a 1% requirement before December 2003, increasing by increments of 0.5% through 2009, and then increase by 1% through 2012, more or less back-loaded schedule.

The real wholesale price of natural gas was projected using the PROSIM model, an economic dispatch model, to be relatively flat starting at \$3.32 per million Btu (MMBtu) in 2003 and rising to only \$3.55 per MMBtu by 2009. As a result of this and the assumption that in the absence of renewables natural-gas-fueled turbines would meet any added demand, average real electricity prices were expected to rise from \$0.0334 per kWh in 2003 to \$0.0355 per kWh in 2012.

Based on these prices, landfill gas in New England is expected, in the base case, to be the main basis for meeting the RPS through 2009 at which point wind power and co-fired biomass will take over, in part because land-fill gas will be near the region's capacity. The real excess cost of renewable electricity generation at the margin is estimated at \$0.0210 per kWh in 2003, rising to \$0.0232 by 2012 per kWh. Of course, after factoring in their relatively low percentage of the fuel set, renewables will increase net customer costs only slightly even by 2012: a 0.4% increase in 2003 to about a 2.2% increase in 2012.

The Massachusetts report goes on to investigate in some detail the effect of uncertainty in the assumptions regarding the costs and, hence, supply of the various renewable technologies. The authors also examine demand-side variations, changes in RPS requirements, variations in retail electricity demand, and changes in non-RPS demand for renewables. Higher-than-expected costs of compliance with the RPS were perceived to drive the added cost premium of wholesale electricity prices as high as \$0.05 per kWh in 2009 and \$0.06 per kWh in 2012. If lower-than-expected costs of compliance occur, then the cost premiums of wholesale electricity prices could be dampened to as low as \$0.01 per kWh over the entire

study period, according to the simulations conducted by the authors

Hawaii

Hawaii is unusual in that it currently relies heavily on oil as an energy resource to produce electric power, which must take place within the island state. Moreover, according to the study by GDS Associates (2001), the delivered price of oil to generation facilities in Hawaii is on the order of 20% higher than the U.S. average to all electric power generators. Plus, as a primary resource, oil leaves Hawaii uniquely vulnerable among the states to the whims of supply disruptions and the environmental risks of oil spills that are almost unique for states of the union. Thus, among all U.S. states Hawaii is a prime location for enhancing its renewable portfolio.

The GDS report analyzed the economic impacts of both a 9.5% and 10.5% RPS for 2010 using two different oil price forecasts (baseline and low). In all scenarios, the percentage of renewable sources was forecast to increase in a straight line from 3% in 2003. Forecast targets of the RPS were broken out by utility. World oil prices were forecast in the baseline case at \$25 per barrel and in the low oil-price case at \$22 per barrel in 2003 well below the current price of oil. In both cases oil prices were forecast to rise at rates established in the Gas Research Institute's 2000 Baseline Projection of oil price growth for its Pacific 2 energy demand region, composed of California and Hawaii. These oil forecasts were broken out into delivered costs by Hawaii electric utility.

While Hawaii has some geothermal and hydroelectric sources that it can and will bring on line by 2010, wind turbines will fulfill the bulk of its new RPS. The projected costs for 2010 per kWh for wind power ranged from \$0.036 at Kohala Wells on the Big Island to \$0.069 at Puunene on Maui. Also no new renewable projects were projected to be installed after 2010. On the other hand, the explicit technology plan laid out in the report made it possible to add in each facility in an order based on the net cost of electricity that each produced – lowest cost first, of course.

Since almost all renewable resources except a few pending solar units produced electric power

at rates below prevailing ones that use fossil fuels in Hawaii, all of the RPS scenarios yielded lower net electric power prices through 2020. Indeed, because of this, high-cost fossil fuel scenarios were not examined. Finally the relative benefits over the alternative of fossil fuel sources of the installation, maintenance, and operation of the renewable power facilities were not considered in the scenario calculations. Also the environmental benefits were not evaluated or for that matter even assessed thoroughly. In part this is because of the prevailing friendliness of Hawaii's economic climate to the use of renewable energy resources.

The net present value of estimated savings under Hawaii's RPS ranged from \$27.8 to \$43.1 million for the period from 2001-2010 and from \$62.4 to \$98.4 million through 2020. While important these costs were deemed small. At their highest, they were only about 2.6% of the total baseline costs of generation over the ten-year period (\$3.2 billion).

Maryland

The Maryland study by Synapse Energy Economics (Chen et al., 2003) presumed that its state would have to meet a 7.5% RPS by 2013. They examined five scenarios, defined by conditions at the end of the study period, the year 2013: (1) natural gas prices rise by 25%, (2) natural gas prices rise by 50%,

Table 1.1 Projected RPS Premium and Percent Change to Maryland's Average Retail Cost of Electricity (\$/kWh)

	2006		2010		2013	
	\$/kWh	% Change	\$/kWh	% Change	\$/kWh	% Change
Baseline	0.0095	0.1	0.0093	0.5	0.0081	0.8
Gas price rises 25%	0.0056	0.0	0.0030	0.1	0.0012	0.1
Gas prices rise 50%	0.0017	0.0	-0.0034	-0.1	-0.0058	-0.4
Gas price falls 2%	0.0098	0.1	0.0098	-0.3	0.0087	-0.6
Gas price falls 4%	0.0101	0.1	0.1029	-1.1	0.0093	-2.0
RPS penalty	0.0200	0.1	0.0200	1.1	0.0200	1.9

Source: Synapse Energy Economics (2003a, Table 6.1)

(3) natural gas prices fall by 2%, (4) natural gas prices fall by 4%, and (5) a \$0.02 per kWh fee for supplier noncompliance with the RPS. In cases (3) and (4), the fall in the delivered price of natural gas was assumed to be due to the application of the new RPS. The

study did not make clear its RPS assumptions for intermediate years.

The costs of renewable energy sources were \$0.045 per kWh for wind turbines running at 34% of capacity; \$0.053 per kWh for landfill-gas fueled turbines; \$0.070 per kWh for direct biomass; and \$0.281 per kWh for solar photovoltaic panels from the Energy Information Administration's *Annual Energy Outlook 2003*. Prevailing 2002 wholesale electricity generation costs in Maryland applied in the study were \$0.035 per kWh. It was deemed to hold at \$0.035 per kWh through 2006 and then increase to \$0.040 per kWh by 2013 in the baseline case, which assumes that all marginal energy in PJM will be produced via natural-gas-fueled turbines. Credit trading within PJM was assumed.

Wind power was supposed to be the first renewable resource of choice due to its lower costs. With that, costs of electricity through the RPS were estimated in the baseline case to rise to \$0.045 per kWh by 2006 and further to \$0.047 per kWh by 2013. The following table summarizes the impacts of the various scenarios on electricity prices in Maryland.

New York

Synapse Energy Economics (Keith et al., 2003) also performed a study for the State of New York. It is strictly a review of pre-existing literature on the subject. It concludes that there are many important benefits of an RPS: (1) emissions reduction programs in the state will provide significant environmental and health benefits, (2) gas prices are volatile and may be higher than average during the coming years, and (3) studies of other areas show investments in renewable power generation yields more and better-quality jobs than do fossil-fuel alternatives. The New York Public Service Commission adopted an RPS of at least 25% by 2013 on September 22, 2004.

Vermont

Synapse Energy Economics' analysis of Vermont's RPS (Woolf et al., 2003) has three RPS scenarios that flow through 2015. The first calls for a new renewable power generation target of 0.5% in 2006 that increases by 0.5 percentage points annually. A second begins at 1% in 2006 with annual 1.0 percentage point increments. The third starts at 2% in 2006 with annual increases of 2.0 percentage points. Natural gas combined-cycle facilities are assumed to be the alternative to renewables for increasing power generation in the future. The future cost of electricity in Vermont was obtained by costing out such a plant and assuming at least one would start operation in 2010. Electricity prices for intermediate years were estimated as linear interpolations.

Vermont is considering whether to include co-fired biomass as eligible under its RPS. If so, it will be the only state in New England to do so. Moreover, the economics of technology makes co-fired biomass among the cheapest electric power fuels for this region. Because of the large number of dairy farms in the state, manure digestion was also deemed economically viable. Hence, in the early years of the Vermont RPS, the report assumes biomass technology will dominate through the expansion of existing co-fired facilities and the installation of manure digesters.

While wind is expected to be a major factor in the supply of power to New England beginning in about 2010, its effect on the supply to Vermont was deemed to be small. The ready availability of the hydroelectric power across the border in Quebec and a better-supported RPS in New York State are expected to make imports to New England important to its Vermont customers. Moreover, regulatory barriers to new hydroelectric dams in New England were perceived to price this option out of economic feasibility. On the other hand, it may be that imports from new facilities in Quebec could be constrained by politics and transmission losses. Hence, extra scenarios were developed by including and excluding this low-cost option in the original one. The scenarios were further modified to consider what would happen if the state's RPS mandated that it pertained to Vermont-based production only or more generally to New England.

In the end, the Vermont report examines the difference between electricity prices that result from the two alternative scenarios to the baseline—that of an RPS with 1.0 percentage point increments. Of course, this is done in each case for three different conditions: (1) assumes the eligibility of low-cost renewables not eligible under the RPS of other New England states (henceforth called, "VT-only eligible") (2) VT-only eligibles but excluding the possibility of importing from Hydro-Quebec, and (3) a New England RPS perspective. Two cases where wholesale electricity prices became 20% lower or higher were also examined.

The impact of the various alternatives on electricity prices was determined to be small, increasing average electricity prices only by 1.5% from the baseline case in 2015 when only a New England eligible 2% annual increase in the RPS was considered. Not surprisingly, the volatility of prevailing electricity prices was deemed more likely to have an impact on the economic viability of a new Vermont RPS than would the various percentages of renewables that were considered. Prices 20% lower than those of the baseline forecast were perceived to make the 1% increment in the RPS enhance average electricity prices by only 1.4% in 2015. On the other hand, 20% higher wholesale prices essentially made the 1%-incremented RPS virtually costless.

Pennsylvania

Pennsylvania's RPS study (Black and Veatch, 2004) is the most recent among member states of PJM. The Pennsylvania RPS is projected to be 1% in 2006 rising a percentage point per year to a total of 10% by 2015 at which point it will level off through 2025. Starting in 2015, biomass co-firing is expected to meet about 21% of the RPS (at an average cost of \$0.0366 per kWh), wind 46% (\$0.0825 per kWh), hydroelectric power 14% (\$0.0576 per kWh), and landfill (\$0.0482 per kWh) and digester gas (\$0.0837 per kWh) fuels combined about 5%. Of course, the capacity mix is somewhat more slanted toward wind power due to the lower capacity utilization of that technology. Solar-photovoltaic-generated electric power was deemed not to be economically viable during the study period.

In the study, the technologies were scheduled by permitting only the lowest-cost alternatives to be built first. Two different wind-power technologies were allotted equal percentages of the “market share,” one costing \$0.07 per kWh and the other \$0.095 per kWh. The report is detailed in its costing of the technologies as well as in its pointing out the respective probable geography of each technology’s installations.

Costs of the RPS were compared to a baseline that assumed the current mix of fossil-fueled plants. Over a 20-year period the RPS was deemed to cost \$1.23 billion or 36 percent more than the baseline case for new installations in present value terms. This, however, equates to an average rate premium of only 0.036 cents per kWh or a 0.46% increase over Pennsylvania’s average retail price of \$0.0786 per kWh in 2001.

Of course, new investments in the RPS can create economic activity in as much as they yield more economic benefits than the baseline alternative. RIMS II multipliers produced by the US Bureau of Economic Analysis were applied to this analysis [1]. Although not clear in the report, the multiplier used are presumably those for Pennsylvania. On a per MW basis, the analysis reveals that wind power and biomass co-firing yield greater annual returns to that state than do conventional technologies. Unfortunately, the analysis mixes numbers of one-time construction impacts with recurring annual impacts from operations and maintenance of the facilities, which makes the summary numbers difficult to interpret. Nonetheless, unlike prior studies at least some effort was undertaken to capture these benefits.

As in prior studies some effort was also made to summarize benefits obtained by possible reductions in the prices of fossil fuels, given that demand for them effectively will decline via the RPS. While not articulated with any detail, the report summarizes that natural gas prices should decline by 3% with what must be assumed to be nationwide RPS-based pressures. Since natural gas maintains but a 3% fuel share in Pennsylvania electric-power generation

(although the percentage of natural gas generation in PJM is substantially larger), the effect of such a small price change would in all likelihood be minimal.

Colorado

Among the most recent reports is one for Colorado (Binz, 2004). It is an investigation into the potential impacts of what Colorado residents call Amendment 37, an initiative on the Colorado ballot in November 2004. This amendment will affect 80% of Colorado electricity consumers, since it pertains to Colorado utilities with 40,000 or more customers. Currently all renewable electric power generation supplies about 1.8% of the power base. The initiative for Amendment 37, which was approved by voters in November 2004, seeks to establish the RPS to 10% by 2025. Moreover, 4% of the RPS (0.4% of all electrical power by 2025) must be met by solar energy sources.

The study uses unsubsidized costs for wind power of energy reported by the US Department of Energy (specific report uncited) of \$0.05 per kWh and falling to \$0.035 by 2023. It admits that this estimate is likely to be conservative based on some anecdotal evidence. Integration and transmission costs add another \$0.008 per kWh. The costs of a central-station solar power from a Sargent and Lundy report for the National Renewable Energy Laboratory are similarly cited. The costs in 2004 are taken to be \$0.14 per kWh and expected to fall to \$0.055 by 2020. These costs are expected to be on the high side, since the report suggests the same study for identified solar technologies that start off costing less and dropping more steeply over the forecast period. The same DOE EIA 2004 forecast for natural gas prices as applied in the present study are used to price the baseline fuel in this one for Colorado. With this information in hand nine scenarios were investigated based upon three different forecasts of natural gas costs and three different possible futures for the federal Production Tax Credit (PTC): no extension, an extension through the end of 2006, and an extension through the end of 2010.

The scenarios were carried out for Xcel Energy, the largest utility in Colorado. Results reveal that, regardless of the scenario, the impacts upon a typical monthly residential utility bill are likely to

[1] RIMS II is a set of state input-output multipliers produced by the U.S. Bureau of Economic Analysis. See Appendix A for more details.)

be very small—at most \$0.63, the case of low gas prices with no extension of the PTC. The aggregate impact over the 20-year period was \$190 million for this same case. Meanwhile certain other cases presented possible economic gains (price drops) with the adoption of renewables. To get a better sense of things, probabilities were assigned to each scenario such that the sum across them was 100%. The net present value of the merging of the scenarios is \$12.6 million in 2004 dollars or about a \$0.01 increase in a typical monthly residential bill.

As in other reports, the RPS was assumed to have significant environmental effects as well. In particular in Colorado water is in many places a scarce resource. Given that it will replace natural gas and coal-fired plants, renewables electricity generation is deemed to reduce water consumption in electricity production by one to two million cubic feet (3.25-6.40 thousand acre-feet) per year.

Chapter 2: Economic Assessment of the Proposed 20% Renewable Portfolio Standard

I. Introduction and Findings

This chapter reports on the incremental economic impacts of the proposed 20% RPS compared to the existing RPS from now until the year 2020. The analysis is based on the R/ECON™ model and additional and more detailed economic analysis of the PV and wind sectors. The R/ECON™ model is a widely accepted economic model that is frequently used to forecast the economic performance of the New Jersey economy. It analyzes growth in such variable as gross state product, employment, income, and prices. It models the economy using a system of econometric equations that have been developed and tested over many years. In addition, it can be used to compare the economic impacts of different policies such as the existing RPS versus the proposed 20% RPS.

This economic analysis starts by forecasting the additional annual cost to the production of electricity due to the proposed 20% RPS compared to the existing RPS from now through the year 2020. To do so, the analysis reviews key national and economic factors and different fossil fuel price projections. (Prior RPS studies of other states have neglected such possible economic repercussions.) The impacts these additional electricity costs have on the New Jersey economy are then determined. Since the proposed 20% RPS, if combined with appropriate economic development policies, could increase employment and associated economic activity in the New Jersey renewable sector, these effects are also quantified.

There are three subsequent sections to this Chapter. Section II describes the R/ECON™ model and its results. It provides the baseline economic assumptions that the

model uses including a high, low and expected fossil fuel price projections. This section also discusses and quantifies the economic implications of two 20% RPS scenarios compared to the existing RPS. The future cost of renewable resources depends substantially upon technological progress, engineering improvements, and learning gained by more and more experience with these technologies. The expected scenario, based on a New Jersey specific renewable market assessment commissioned as part of this project, is that these costs decline faster than they have in the recent past in New Jersey.

An alternative scenario is that renewable costs decline as they have done historically. Section III reports an additional and more detailed economic assessment of the proposed 20% RPS on the New Jersey PV and wind industries. To the extent that construction, operation, and maintenance facilities and personnel are located in New Jersey, the proposed 20% RPS would provide additional jobs and economic activity in these areas compared to the existing RPS. Section IV concludes this chapter.

This chapter finds the following:

1. Under the expected case assumptions, the proposed 20% RPS compared to the existing RPS would raise electricity prices approximately 3.7% in the year 2020 and have no measurable impact on the growth of New Jersey's economy;
2. If natural gas prices rise to levels assumed in the High Energy Price scenario, the proposed 20% RPS has a positive economic impact on the New Jersey economy because electricity

prices would be lower than under the existing RPS scenario;

3. The economic and electricity price impacts of the proposed 20% RPS depend substantially on whether expected technological improvements occur that reduce the cost of PVs and wind power;
4. Under the proposed 20% RPS, the location in New Jersey of all of the manufacturing, operations, and maintenance facilities and employees needed to support the PV and wind infrastructure, for instance if New Jersey developed its off-shore wind capability or if the regional wind infrastructure is located in New Jersey, would add approximately 11,700 jobs and attenuate economic benefits to the New Jersey economy between 2008 and 2020;
5. The proposed 20% RPS would lower natural gas prices consumers pay by reducing the burning of this fuel in power generation; and
6. The PV portion of the proposed 20% RPS would increase reliability by providing electricity when grid power is not available and may reduce expenditures on T&D within the state.

II. Description of the R/ECON™ Model and its Results

The Rutgers Economic Advisory Service Econometric Model of the New Jersey Economy (R/ECON™) is an econometric model comprised of over 150 equations the solutions of which are solved simultaneously. The equations are based on historical data for New Jersey and the US. The historical data used to produce the model covers the period from 1970 to the second quarter of 2003. The sectors included in the model are:

- Employment and gross state product for 40 industries
- Wage rates and price deflators for major industries
- Consumer price index
- Personal income and its components
- Population, labor force and unemployment
- Housing permits and construction contracts
- Motor vehicle registrations, and
- State tax revenues by type of tax, and current and capital expenditures.

The heart of the model is a set of equations modeling employment, wages, and prices by industry. In general, employment in an industry depends on demand for that industry's output, and on the state's wages and prices relative to the nation's wages and prices. Demand can be represented by a variety of variables including (but not limited to) New Jersey personal income, population, or US employment in the sector. Demand for retail trade is represented by published New Jersey retail sales [2]. For the construction sector it is represented by the value of construction contracts. Growth in population is driven by total employment in the state and by state wages and prices relative to national wages and prices.

As part of this project the model was extended to include several equations related to the energy sector. The equations in this new model sector are:

- electric price per kilowatt hour, residential and other (commercial and industrial);
- electricity usage per 1000 megawatt hours, residential;
- electric revenues in billions of dollars residential and other;
- energy taxes in millions of dollars, sales and corporate business; and
- employment at electric utilities and other utilities [3].

To be consistent with this classification system, *the term utility used in this chapter also encompasses the generation sector of the supply chain and not just the traditionally regulated transmission and distribution functions.* As used in this report utilities include “electric power, natural gas, steam supply, water supply, and sewage removal,” as per the industrial definition in the North American Industry Classification System [4]. The employment data for

[2] State retail sales were published on a monthly basis by the US Department of Commerce until 1995. Later data is estimated by R/ECON™.

[3] The employment data, like all other New Jersey employment data used in the model, comes from the New Jersey Department of Labor.

[4] North American Industry Classification System, p. 85.

electrical utilities includes jobs in “establishments primarily engaged in generating, transmitting, and/or distributing electric power” [5].

Other information added to the model for this project includes variables showing the fraction of electricity used in New Jersey that is produced by renewable resources, both PVs and other renewable sources. This fraction influences the price of electricity.

The data for electric prices, usage, and revenues were developed from monthly data for 1990 to current supplied by four utilities operating in New Jersey: PSE&G, JCP&L Energy, Connectiv, and Orange and Rockland. The reports from the utilities contained megawatts used and revenues per month by type of user including: residential heat and other, commercial firm and other, industrial firm and other, other, and sales for resale. The megawatts and revenues per month were aggregated to total residential and total other. Electricity prices were calculated as revenue per month divided by megawatts per month for total, residential, and other.

Energy taxes were developed from the monthly collections data [6] for the period from 1998 to current. The taxes include corporate business and sales taxes for energy and Transitional Energy Facilities Assessment (TEFA). The TEFA is assumed to phase out through FY2006 and then not be renewed.

Besides adding these equations and data to the model other changes were made to incorporate output from the new equations into other parts of the model. For instance, the equation estimating the consumer price index for New Jersey was modified to include the price of electricity for residential use.

The Baseline (BASELINE) Forecast

R/ECON™ produces four forecasts each year of the New Jersey economy using the econometric model. This study used the April 2004 R/ECON™ forecast, modified to include the new energy sector, as its baseline (referred hereafter as BASELINE). The baseline forecast goes out to 2020. It is the underlying economic forecast of macroeconomic variables for this study. The data for the US used in BASELINE comes from the Global Insight, Inc. forecast of February 2004. Global Insight, Inc. is a national leader in economic forecasting and is used in other studies conducted at the Bloustein School [7].

Overview of the US Forecast

The Global Insight February 2004 forecast looks for slow, steady growth after 2005, with slower

Table 2.1 Summary of U.S. Economic Forecast, 2003 to 2020

	2003	2004	2005	2005 to 2020
Annual Percentage Growth				
Nonagricultural Employment	-0.3%	1.2%	2.1%	1.1%
Real Domestic Product	3.1%	4.9%	3.8%	3.0%
Personal Income	3.3%	5.7%	5.6%	5.9%
Consumer Price Index	2.3%	2.7%	1.5%	2.4%
Producer Price Index: Energy	21.2%	9.6%	-5.3%	1.0%
Percentage				
Unemployment Rate (year)	6.0%	5.5%	5.3%	4.7%

Source: Global Insight, *U.S. Economic Outlook*, February 2004.

growth in output and employment than in the past two decades, but also less inflation. Real gross domestic product (GDP) growth of 4.9% in 2004 is the strongest performance during the forecast period. Growth will return to a more sustainable rate in 2005, and average 3% a year from 2005 to 2020, somewhat lower than the average of 3.3% per year experienced over the past 2 decades. (See Table 2.1.)

Nonagricultural employment will rise by 1.2% in 2004. Peak growth in the cycle will occur in 2005, and then job growth will average 1.1% a year. This

[5] North American Industry Classification System, p. 85.

[6] Comparative Collection Summary, 1998 to present.

[7] More information can be found at Global Insight, Inc. web-site: <http://www.globalinsight.com/>.

long-term average growth rate is considerably lower than the 1.8% average growth rate experienced over the past two decades. The unemployment rate will fall to 5.5% in 2004, 5.3% in 2005, and will average 4.7% between 2005 and 2020.

Consumer prices are expected to rise 2.7% this year, fall back to 1.5% in 2005 with declining oil prices, and average 2.4% a year from 2005 to 2020. The producer price index for energy (PPI Energy) is expected to rise 9.6% in 2004 after an even greater rise in 2003. According to this forecast, it will fall with declining oil prices in 2005 and the increase at an average rate of 1% a year over the rest of the forecast period. As of the writing of this report, oil prices are at an all-time high of \$50 per barrel. The PPI Energy forecast, which was developed prior to the recent run-up in oil prices, is used in order to have an internally consistent set of energy and other economic assumptions. Different fossil fuel price projections from the ones that form the BASELINE forecast are also considered and discussed below.

Overview of the Forecast for New Jersey

The forecast for the state is similar to that for the nation. Nonagricultural employment is expected to

increase 1.3% this year and at an average rate of 1.2% a year for the rest of the forecast period. The growth in real gross state product will rise to 3% this year as the economy expands strongly, fall back slightly in 2005, and then increase at an average of 3.1% a year in the rest of the forecast period. Consumer prices [8] are expected to rise a bit more slowly than in the US in the forecast period. The unemployment rate will fall to 5.2% in 2004 and 2005 and then fall during most of the rest of the forecast period, averaging to 4.9% from 2005 to 2020. (See Table 2.2.)

BASELINE for New Jersey includes the assumption that the Renewable Portfolio Standard will conform to current regulations. That is, Class 1 renewables in 2004 (calendar year) will be 0.75%, rising to 4% in 2009 and remaining at 4% through 2020. Class 2 renewables will remain at 2.5% throughout the period. (Summaries of the BASELINE forecast and other scenarios can be found in Appendix A.)

In BASELINE electric prices rise by 8% from \$0.10 per kilowatt-hour in 2004 to \$0.108 in 2020. The small rise is not surprising since the producer price index for energy declines sharply between 2004 and 2008, and is only 11% higher in 2020 than in 2004. Over the 2004 to 2020 period electric usage rises by 36% and electric revenues by 46%. Total taxes attributable to electric utilities fall slightly because of the TEFA phase-out, but exclusive of TEFA they rise by 24%. Table 2.3 summarizes these figures.

Economic Analysis of the Impact of Other Energy Producer Price Scenarios on the New Jersey Economy

In BASELINE for the US and New Jersey, the producer price index for energy (PPI Energy) for fuels, related products, and power, rose from 1.24 [9] in 2004 to 1.37 in 2020. That 10.5% growth, however,

Table 2.2 Summary of New Jersey Economic Forecast, 2003 to 2020

	2003	2004	2005	2005 to 2020
Annual Percentage Growth				
Nonagricultural Employment	-0.1%	1.3%	1.2%	1.2%
Real Gross State Product	2.1%	3.0%	2.7%	3.1%
Personal Income	3.2%	5.4%	5.0%	5.6%
Consumer Prices	2.5%	2.0%	1.4%	2.3%
Percentage				
Unemployment Rate(year)	5.9%	5.2%	5.2%	4.9%

Source: R/ECONtm

[8]The consumer price index for New Jersey is calculated for the purposes of the R/ECONtm as a population weighted average of the New York-New Jersey-Connecticut CPI and the Pennsylvania-New Jersey CPI.

[9] The base year for the producer price index is 1982.

Table 2.3 New Jersey Electric Utilities: Prices, Usage, and Taxes

	2004	2020	Growth 2004-2020
Electric Price per Kilowatt Hour	\$0.100	\$0.108	8%
Electricity Usage in 1000 Megawatt Hours	66,537	90,539	36%
Electric Revenues (\$ Billions)	\$6.66	\$9.85	48%
Energy Taxes (\$ Millions)	\$1,108.3	\$1,060.3	-4%
Energy Taxes - TEFA (\$Millions)	\$858.8	\$1,060.3	23%

masks a decline of 14.9% during the period from 2004 to 2008. The coal component of the index increased 9.1% over the 16-year period with a minor decline in 2005, while the natural gas component increased 5.3% over the period with a decline of 19.8% between 2004 and 2008. The top half of Table 2.4 presents energy price indices, and its lower half presents the annual escalation rates.

In an attempt to see what will happen to the New Jersey economy if energy prices rise either more or less than in the BASELINE, two energy price scenarios were created. In the first (Low Energy Price) the producer price index for coal (PPI Coal) remains fixed from 2004 to 2020 and the producer price index for natural gas (PPI NG) increases 1% a year beginning in 2009. In the second (High Energy Price) PPI Coal increases 1% a year beginning in 2005 and PPI NG increases 3% a year beginning in 2005. The growth rates of all other components of the PPI

Energy remain as they were in the BASELINE forecast. Figures 2.1 through 2.3 show the PPIs for coal, natural gas, and energy for the baseline, Low Energy Price, and High Energy Price assumptions. These alternative scenarios were formulated from a review of other fuel forecasts (EIA, 2004; National Petroleum Council, 2003).

Because of the decline in PPI NG between 2004 and 2008 in BASELINE we begin to let PPI NG grow at 1% beginning in 2009. Otherwise, the level of PPI NG in the low growth scenario would be higher than in BASELINE.

A comparison, shown in Table 2.5, of the high growth PPI Energy scenario with the BASELINE shows very little difference between the two. In the High Energy Price scenario, electric prices in New Jersey are 2.9% higher than in BASELINE by 2010 and 4.6% higher in 2020. This leads to a projected reduction in electricity usage of .04% in 2010 and .02% in 2020, and an increase of 2.3% in electric revenues in 2010 and 4.9% in 2020. Projected state tax revenues from electric companies would be 1.4% higher in 2010 and 2.6% higher in 2020. There would be a small decline in gross state product at utilities and a negligible decline in overall gross state product and

Table 2.4 Annual Wholesale Energy Price Indices and Escalation Rates for the US

Baseline																	
	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
Total	1.24	1.17	1.08	1.06	1.05	1.06	1.08	1.12	1.14	1.17	1.20	1.22	1.25	1.28	1.31	1.34	1.37
Coal	1.05	1.04	1.04	1.07	1.08	1.08	1.08	1.09	1.10	1.10	1.11	1.12	1.12	1.13	1.13	1.14	1.15
Natural Gas	1.86	1.81	1.61	1.55	1.49	1.51	1.55	1.62	1.65	1.71	1.75	1.78	1.81	1.85	1.88	1.92	1.96
Total	9.6%	(5.3%)	(8.0%)	(1.6%)	(0.7%)	0.9%	1.9%	3.1%	2.2%	2.4%	2.3%	2.3%	2.3%	2.3%	2.3%	2.3%	2.3%
Coal	5.1%	(1.1%)	0.4%	2.4%	0.7%	0.1%	0.8%	0.7%	0.6%	0.5%	0.5%	0.4%	0.5%	0.5%	0.5%	0.5%	0.5%
Natural Gas	8.4%	(2.7%)	(11.1%)	(3.8%)	(3.6%)	1.4%	2.7%	4.6%	2.0%	3.3%	2.3%	1.9%	1.9%	1.9%	1.9%	1.9%	1.9%

Figure 2.1 PPI for Coal, 2004 to 2020

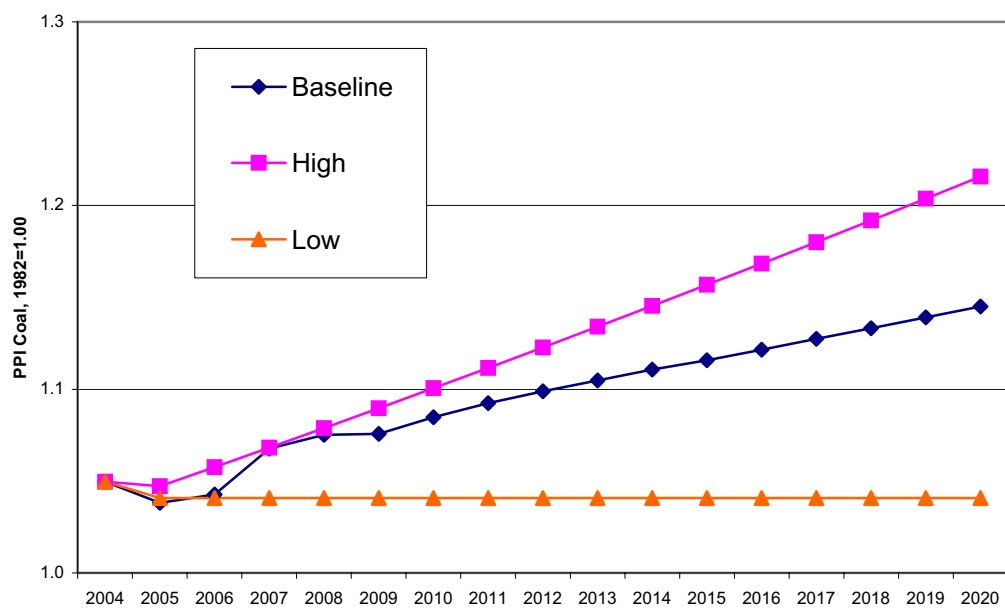


Figure 2.2 PPI for Natural Gas, 2004 to 2020

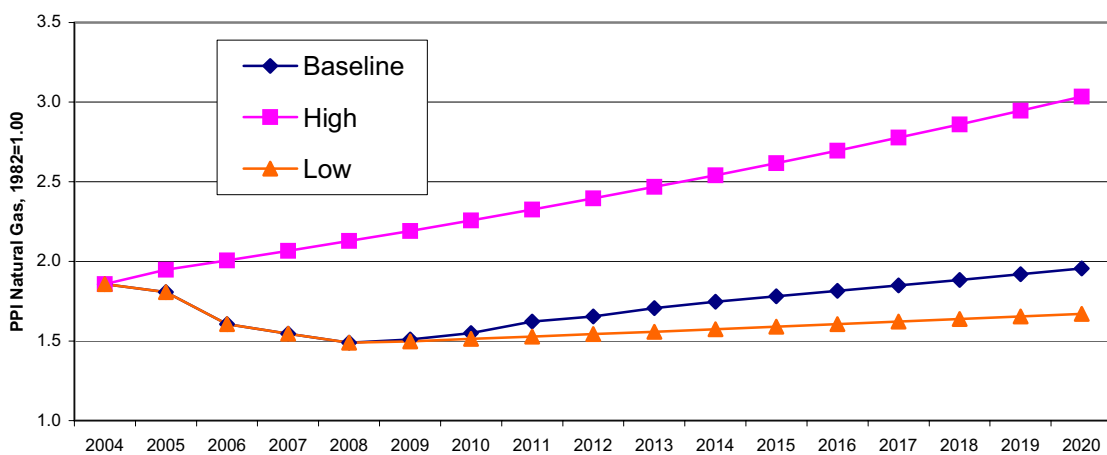


Figure 2.3 PPI for Energy, 2004 to 2020

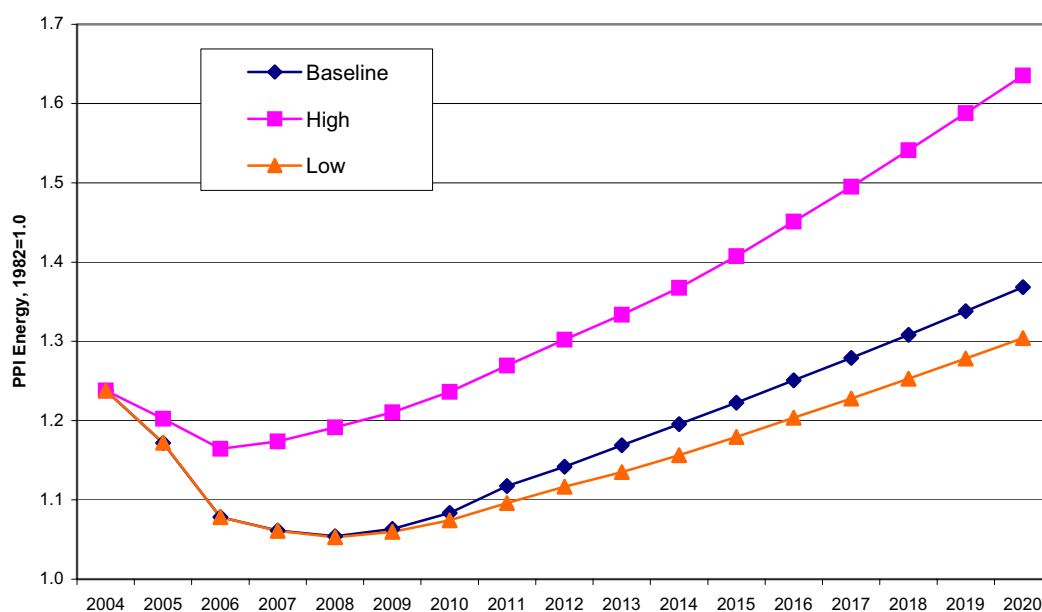


Table 2.5 Comparison of High Growth PPI Energy vs. BASELINE

Comparison Points	Percentage Difference		Total Difference 2004 to 2020*
	2010	2020	
Electric Price per Kilowatt Hour	2.941%	4.630%	
Electricity Usage in 1000 Megawatt Hours	-0.037%	-0.022%	-403.6
Electric Revenues (\$ Millions)	2.262%	4.912%	\$4,247.8
Energy Taxes (\$ Millions)	1.422%	2.559%	\$241.9
Gross State Product for Utilities (\$ Millions 2000=100)	-0.213%	-0.045%	-\$0.2
Gross State Product (\$Millions 2000=100)	-0.053%	-0.039%	-\$3.2
Employment at Electric Utilities (Thousands)	-0.026%	-0.033%	-0.05
Nonagricultural Employment (Thousands)	-0.012%	-0.011%	-7.53
Consumer Price Index (1982=100)	0.132%	0.103%	

*All total differences reported in this chapter are the sum of the annual values and are not discounted.

non-agricultural employment.

The differences between the Low Energy Price scenario and the BASELINE are even smaller than between the High Energy Price scenario and the BASELINE because there is less difference in the

additional scenario of back loading the proposed 20% RPS, i.e., having a larger percentage of the proposed 20% RPS requirement occur in later years, was also analyzed, but there was only a slight difference in its impact when compared to the straight-line scenario. The next section, III.D, assumes that there is no

change in the technology beyond the trends embedded in the model of providing renewable sources of energy.

The increase in the RPS is expected to increase the price of electricity for both industrial

Table 2.6 Comparison of Low Growth PPI Energy vs. BASELINE

Comparison Points	Percentage Difference		Total Difference 2004 to 2020
	2010	2020	
Electric Price per Kilowatt Hour	0.000%	-0.926%	
Electricity Usage in 1000 Megawatt Hours	0.008%	0.010%	120
Electric Revenues (\$ Millions)	-0.614%	-1.491%	-\$0.7
Energy Taxes (\$ Millions)	-0.022%	-0.654%	-\$38.3
Gross State Product for Utilities (\$ Millions 2000=100)	0.045%	0.027%	\$0.1
Gross State Product (\$Millions 2000=100)	0.016%	0.013%	\$1.1
Employment at Electric Utilities (Thousands)	0.007%	0.013%	0.02
Nonagricultural Employment (Thousands)	0.004%	0.004%	2.51
Consumer Price Index (1982=100)	-0.043%	-0.034%	

levels of the PPI Energy indexes presented in chart 3. Table 2.6 summaries these differences.

Economic Analysis the Proposed 20% RPS with Expected Cost Reductions

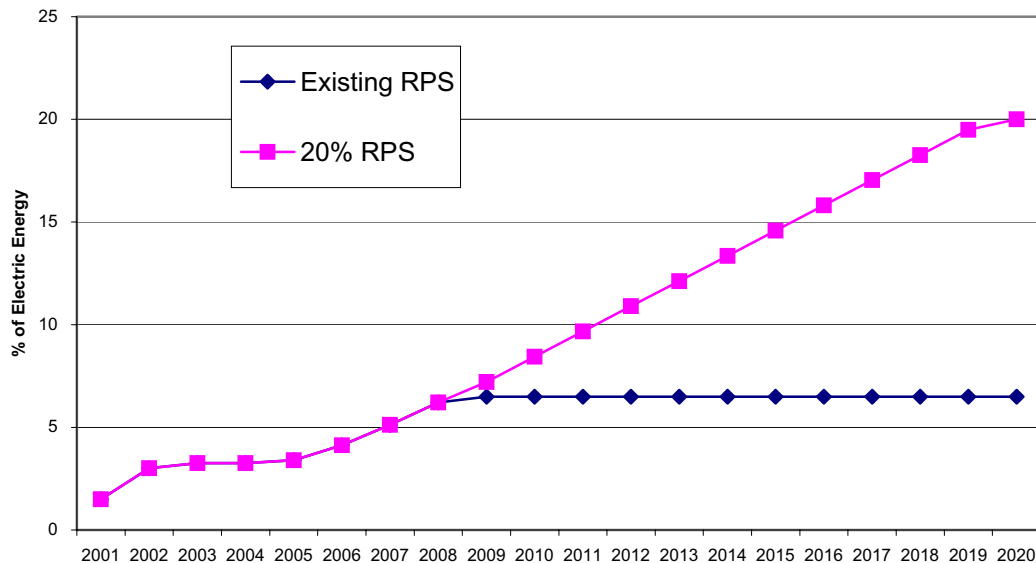
This section compares the economic impact of the proposed 20% RPS with the existing RPS, assuming expected cost reductions in wind and PVs. Under the current rules the renewable portfolio standard (RPS) will rise to 6.5% in 2008 (BPU year) or 2009 (calendar year) [10]. A simulation of the R/ECON™ model called 20% RPS Expected Cost Reductions shows the economic impacts of increasing the Class 1 RPS to 20% by 2020 (calendar year). The increases in the RPS are assumed to occur on a straight-line basis (that is by equal amounts each year including for PVs so that PVs are the same proportion as Class 1 renewables in the year 2020 as in 2008) from 2009 to 2020. Figure 2.4 compares the RPS in the BASELINE and proposed 20% RPS scenarios. An

and residential use. In general, the increase in electric prices is expected to lead to a decrease in usage and also to a decrease in economic activity in the state. Whether the increase in the RPS would increase or decrease corporation and sales tax revenues from the utilities depends on the elasticity of the demand for electricity. Since electricity demand is relatively price inelastic for the small price changes explored in this report, sales tax revenues should not change substantially due to the reduction in electricity sales caused by higher electricity prices.

The data used in this section derives from a New Jersey Market Assessment for renewable energy commissioned as part of the Comprehensive Resource Analysis proceeding conducted by the BPU (Navigant Report, 2004). The purpose of the proceeding is for the BPU to set the appropriate level of funding for energy efficiency and renewable programs. These advances in technology, engineering improvements, and learning gained by more and more experience with these technologies are expected to result in a considerable decrease in the price of electricity provided by renewable resources, although the price

[10] The RPS program year starts on June 1 and ends on May 31 with the program year being the year as of June 1.

Figure 2.4 Renewable Portfolio Standard, 2001 to 2020



would still be higher than it would be using coal, oil, and gas fired facilities. To calculate this additional cost, the market clearing prices for PV and Class 1 RECs are forecasted. In the case of PVs, the REC price is set at the difference between the cost of PVs and the wholesale price for electricity, which was determined as part of the New Jersey Market assessment by a regional economic dispatch model. The price of the Class 1 REC is set at the difference between the cost of the marginal Class 1 resource and the wholesale price of electricity. The marginal Class 1 resource is the most expensive Class 1 resource that is used to satisfy the Class 1 requirement. For the years 2009 and beyond, the marginal Class 1 resource is wind. Although some of the Class 1 requirement that year would be satisfied by landfill gas, biomass and other Class 1 resources, wind power is necessary to meet the requirement. As a result, the price of the Class 1 REC must be sufficient to attract wind resources. Table 2.7

shows the calculation of the increase in electricity prices in 2010, 2015 and 2020 [11].

The first analysis shows the impact of the proposed 20% RPS Expected Cost Reductions on energy prices. (See Table 2.8) Later simulations will add renewable resource facilities construction and maintenance to the analysis.

The slight increase in the price of electricity is forecasted to result in a decline in electricity usage, as well as on gross state product and employment. However, since the decline in usage is very small and demand for electricity is relatively inelastic, the revenues of the utilities and energy taxes increase slightly.

The scenario of higher energy prices showed above that higher producer prices for energy would raise electric prices and reduce electric usage, as well as output and employment in the state. When the proposed 20% RPS Expected Cost Reductions scenario is compared to the High Energy Price scenario, Table 2.9 shows that electricity prices would be lower but electricity usage would not be consistently higher. Overall, however, this proposed 20% RPS scenario

[11] Since the Class 2 requirement is assumed to be the same under the existing and 20% RPS scenarios, there is no difference in costs between them and therefore no difference in economic impact.

Table 2.7 Summary of the Calculation of the Increase in Electricity Prices between the Existing RPS and Proposed 20% RPS, Selective Years (2004 Dollars)

	<u>2010</u>	<u>2015</u>	<u>2020</u>
	Existing RPS (6.5%)		
Cost of PV (\$/MWh)	\$504.29	\$390.00	\$31.00
Cost of Wind Resource (\$/MWh)	\$82.50	\$75.00	\$70.00
PV (GWh = 000's MWh)	127.4	135.9	144.9
Class I (GWh)	3,274.7	3,493.4	3,723.1
PV Net Shortfall (millions)	\$58	\$44	\$35
Class I Net Shortfall (millions)	\$124	\$72	\$54
Total Net Shortfall for Existing RPS (millions)	\$181	\$117	\$89
	20% RPS		
PV (GWh = 000's MWh)	147.5	327	724.5
Class I (GWh)	3,790.8	8,405.2	18,619.5
PV Net Shortfall (millions)	\$67	\$107	\$176
Class I Net Shortfall (millions)	\$143	\$174	\$270
Total Net Shortfall for 20% RPS (millions)	\$210	\$281	\$446
Shortfall Difference Between 20% and Base RPS (millions)	\$29	\$164	\$357
Shortfall Different Between 20% and Base RPS (\$/kWh)	\$0.0004	\$0.0019	\$0.0039
Base Case Electricity Price (\$ per kWh)	\$0.102	\$0.105	\$0.108
20% RPS Case Electricity Price (\$ per kWh)	\$0.102	\$0.107	\$0.112
% Change in Electricity Price Between 20% and Existing RPS	0.4%	1.8%	3.7%

Table 2.8 Proposed 20% RPS Assuming Cost Reductions beyond Historical Trends vs. Existing RPS

Comparison Points	Percentage Difference		Cumulative Difference 2004 to 2020
	2010	2020	
Electric Price per Kilowatt Hour	0.000%	3.704%	
Electricity Usage in 1000 Megawatt Hours	-0.032%	-0.031%	-408.41
Electric Revenues (\$ Millions)	-0.031%	3.307%	\$1.9
Energy Taxes (\$ Millions)	0.118%	1.964%	\$123.3
Gross State Product for Utilities (\$ Millions 2000=100)	-0.001%	0.000%	\$0.0
Gross State Product (\$Millions 2000=100)	-0.001%	-0.015%	-\$0.7
Employment at Electric Utilities (Thousands)	0.000%	-0.007%	-0.01
Nonagricultural Employment (Thousands)	-0.001%	-0.005%	-1.68
Consumer Price Index (1982=100)	0.035%	0.035%	

would improve the economy in terms of output and employment, but utility revenues and taxes would be slightly lower. This analysis demonstrates that if fuel prices rise to levels beyond those assumed in the BASELINE assumptions, the proposed 20% RPS's economic impact declines. The High Energy

Price scenario is approximately the breakeven point in which the proposed 20% RPS Expected Cost Reductions scenario provides positive impact on the economy compared to the renewable levels in the existing RPS. The above analysis does not include any

Table 2.9 Proposed 20% RPS vs. Existing RPS Assuming High Energy Price Scenario

	Percentage Difference		Cumulative Difference 2004 to 2020
	2010	2020	
Electric Price per Kilowatt Hour	-2.857%	-0.885%	
Electricity Usage in 1000 Megawatt Hours	0.005%	-0.009%	-150
Electric Revenues (\$ Millions)	-2.242%	-1.530%	-\$2.0
Energy Taxes (\$ Millions)	-1.286%	-0.580%	-\$105.7
Gross State Product for Utilities (\$ Millions 2000=100)	0.212%	0.045%	\$79.5
Gross State Product (\$Millions 2000=100)	0.051%	0.024%	\$2,065.0
Employment at Electric Utilities (Thousands)	0.035%	0.048%	0.04
Nonagricultural Employment (Thousands)	0.011%	0.007%	4.89
Consumer Price Index (1982=100)	-0.097%	-0.067%	-109.35

positive economic impact due to additional economic activity from the proposed 20% RPS or the benefits from avoiding environmental emissions.

Economic Analysis of the 20% RPS with Historical Cost Reductions

Figure 2.5 compares the price of electricity in cents per kilowatt-hour in BASELINE and the proposed 20% RPS scenarios. In the Historical Cost Reduction scenario, the price difference from the

model assumes no improvements in technology, engineering, or from experience occur from that incorporated in the data in the model. This historical cost reduction scenario is therefore a “worst case” scenario because all of the sources of further cost reductions are ignored.

Table 2.10 shows a comparison of the two 20% RPS scenarios (Expected versus Historical Cost Reductions) in years 2010 and 2020 as well as the

Figure 2.5 Comparison of Electricity Prices under the proposed 20% RPS-Expected Cost Reductions, 20% RPS-Historical Cost Reductions, and Existing RPS

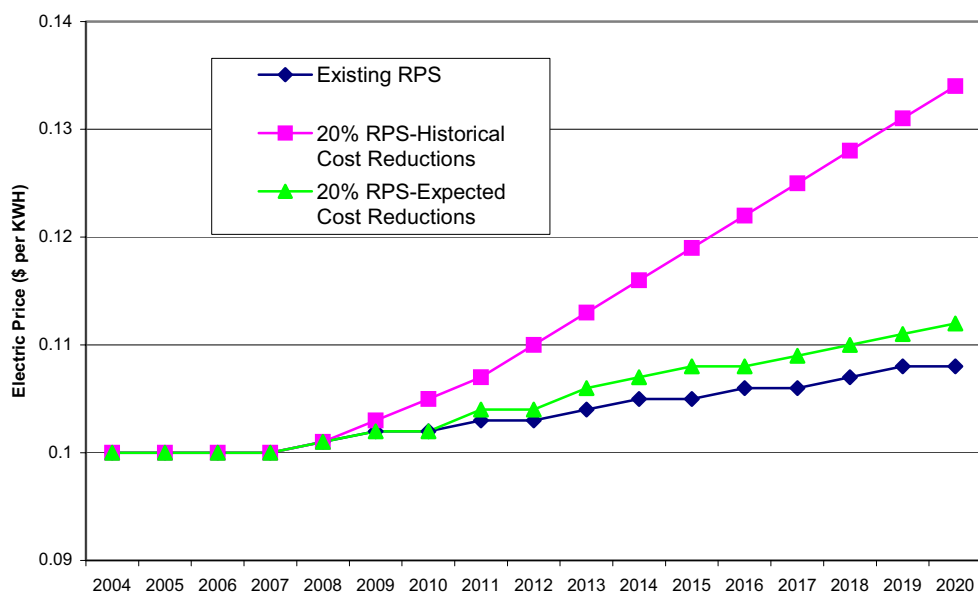


Table 2.10 Existing RPS vs.20% RPS-Historical Cost Reductions

	Percentage Difference 2010	Percentage Difference 2020	Cumulative Difference 2004 to 2020
Electric Price per Kilowatt Hour	2.941%	24.074%	
Electricity Usage in 1000 Megawatt Hours	-0.149%	-0.105%	-1,652
Electric Revenues (\$ Millions)	1.448%	22.660%	\$12,704.4
Energy Taxes (\$ Millions)	1.209%	13.036%	\$785.2
Gross State Product for Utilities (\$ Millions 2000=100)	0.002%	-0.070%	-\$69.5
Gross State Product (\$Millions 2000=100)	-0.008%	-0.145%	-\$4,763.9
Employment at Electric Utilities (Thousands)	-0.007%	-0.086%	-0.05
Non Agricultural Employment (Thousands)	-0.003%	-0.042%	-10.70

total difference in each of the variables over the period from 2004 to 2020 [12].

The expectations regarding the direction of change in the variables mentioned above are borne out in the comparison in the historical cost reduction scenario. The increase in the RPS to 20% by 2020 increases the price of electricity by 2.9% in 2010 and by 24.1% in 2020. The increase in price leads to a small decrease in usage adding up to 1,652 megawatts between 2009 and 2020. Revenues to the utilities increase over the years as much as 22.7% by 2020 and energy taxes increase by 13% by 2020.

In real terms (that is, in constant 2000 dollars) gross state product to the utilities falls slightly as does total gross state product. By 2020 real gross state product is 0.145% lower than it would be if the RPS were not increased. Employment in the state would also be slightly lower. By 2020 the number of jobs in the economy would be 2,000 (0.042%) less than if the RPS were not increased. Chapter 4 discusses the policy implications of the sensitivity of the impact on electricity prices and economic growth due to technological improvements and associated cost reductions in PVs and wind power.

Detailed Economic Assessment of New Energy Resources in New Jersey

As part of this study the R/ECON™ Input-Output model (I-O) was used to estimate the impact of building various kinds of new electric energy resources in New Jersey. This I-O model allows a more detailed economic impact comparison of the PV and wind sectors between the proposed 20% RPS and the existing RPS. This section of the report considers the economic impact of building and maintaining wind driven power plants in New Jersey. This could occur if New Jersey develops off-shore wind or if the bulk of the facilities needed to support wind are located in New Jersey even if the windmills are located in other states, and separately, at the impact of manufacturing and installing photovoltaic systems in

New Jersey. The systems would be built on a schedule conforming to the need to put new systems in place to meet the proposed 20% RPS requirement when the Class 1 requirement is assumed to go from 4% in 2009 to 20% in 2020 on a straight-line method.

Table 2.11 Solar PV and Wind Installations Under the proposed 20% RPS

	Solar PV (MWs Installed)		Wind (MWs Installed)	
	Cumulative	Annual	Cumulative	Annual
2005	6	6		
2006	11	5		
2007	26	15		
2008	56	29	266	266
2009	110	55	513	248
2010	130	19	685	172
2011	152	22	886	201
2012	178	26	1122	236
2013	209	31	1399	277
2014	245	36	1725	326
2015	287	42	2125	399
2016	337	50	2539	414
2017	395	58	3023	484
2018	463	68	3590	567
2019	542	80	4256	666
2020	636	94	4864	608

Table 2.11 shows the number of 8 megawatt solar photovoltaic plants and 60 megawatt wind installations necessary to get to the proposed 20% RPS.

The results from I-O were entered into the R/ECON™ econometric model to get at the long-term impacts of building the renewable facilities in New Jersey. For the purposes of working with the econometric model, it was assumed that the total impacts shown in I-O for each unit put in place would

[12] The differences between STRAIGHTLINE and BASE-LINE all occur between 2009 to 2020. The heading of the last column in the table is for 2004 to 2020 to keep it consistent with the rest of the difference tables in this chapter of the Report.

[13] The results were entered into the econometric model by increasing employment in an industry by the amount of change shown for that industry in the I-O model. Where the I-O change was very small (less than 10) no attempt was made to include it specifically.

occur in 1 year. Thus, if two facilities should be built in a year, the total impact of one unit from I-O would be multiplied by two before adding the impacts into the econometric model [13].

Table 2.12 shows the impact of the 20% RPS standard on the state's economy when the new energy sources are not manufactured in New Jersey. Essentially this simulation shows the impact of installation of PV cells in New Jersey, when they are manufactured elsewhere. This is an extension of Table 2.8 which showed only the impact of the 20% RPS on prices without indicating how the energy would be supplied. In this case, there is an increase in the price of electricity of 3.7 percent in 2020, and a small decrease in total electricity usage of 0.016 percent in 2010 and 0.004 percent in 2020. Revenues to the electrical utility sector would increase by 3.334 percent in 2020, and energy taxes would increase by 0.13 percent in 2010 and 1.982 percent in 2020. It

would decrease jobs in the utility sector very slightly while adding an average of about 700 jobs a year to total employment.

Table 2.13 shows the impact of manufacturing and installing photovoltaic and wind energy facilities in New Jersey according to the schedule shown in Table 2.11, as well the impact of maintaining the systems. The addition of a total of 636 MWs of photovoltaic facilities and 4,864 MWs of wind power as sources of electricity in New Jersey over the period from 2005 to 2020 would increase total electricity usage by 0.071 percent in 2010 and 0.159 percent in 2020. It would increase revenues to this sector by 0.052 percent in 2010 and 3.493 percent in 2020, and increase energy taxes by 0.056 percent in 2010 and .105 percent in 2020. It would add an average of 10 jobs a year to the industry job base while adding an average of about 5,700 jobs a year to total employment. The increase in total jobs would be about 750 in 2005

Table 2.12 20% RPS-Expected Changes with New Energy Sources from outside New Jersey vs. BASELINE

	Percentage Difference		Total Difference
	2010	2020	2004 to 2020
Electric Price per Kilowatt Hour	0.000%	3.704%	
Electricity Usage in 1000 Megawatt Hours	-0.016%	-0.004%	-254.86
Electric Revenues (\$ Millions)	-0.02%	3.334%	\$1.96
Energy Taxes (\$ Millions)	0.130%	1.982%	\$124.50
Gross State Product for Utilities (\$ Millions 2000=100)	0.006%	0.007%	\$0.00
Gross State Product (\$Millions 2000=100)	0.004%	-0.005%	-\$0.35
Employment at Electric Utilities (Thousands)	0.000%	-0.024%	-0.01
NonAgricultural Employment (Thousands)	0.012%	0.037%	10.78
Consumer Price Index (1982=100)	0.036%	0.036%	

Table 2.13 20% RPS-Expected Improvements with PVs and Wind Facilities Manufactured and Maintained in New Jersey vs. 20% RPS-Expected Cost Reductions without These Changes

	Percentage Difference		Total Difference
	2010	2020	2004 to 2020
Electric Price per Kilowatt Hour	0.000%	0.000%	
Electricity Usage in 1000 Megawatt Hours	0.071%	0.159%	1,060
Electric Revenues (\$ Millions)	0.052%	3.493%	\$0.11
Energy Taxes (\$ Millions)	0.056%	0.105%	\$8.38
Gross State Product for Utilities (\$ Millions 2000=100)	0.037%	0.031%	\$0.03
Gross State Product (\$Millions 2000=100)	0.074%	0.160%	\$7.64
Employment at Electric Utilities (Thousands)	0.000%	0.000%	0.2
Non Agricultural Employment (Thousands)	0.081%	0.236%	86.8
Consumer Price Index (1982=100)	0.000%	0.001%	

The increase in total jobs would be about 750 in 2005 and 2006 and rise to about 11,500 in 2020 when 702 MWs of new capacity would be needed, on top of the increased activity in the state from previous building, installation, and maintenance activities,

Since the R/Econ econometric model is a dynamic model, it assumes that the multiplier effects of the RPS technology will occur for some years following their installation. Alternatively, by virtue of its static nature, the I-O model assumes that all multiplier effects will occur in the year of installation. Thus there is likely to be some difference in the economic impacts estimates that emanate from the two models. In the case of the base scenario where no economic development scenarios are available for NJ-based manufacturing of RPS technology (see Table 12.14), therefore, the total number of jobs created from the installation of PV to meet the 20% RPS is expected to peak at about 2,600 by 2020. In 2020, these jobs would support \$191 million in annual total wealth, including \$142 in job earnings and \$12.9 million in state and local tax revenues in year 2000 dollars.

In meeting New Jersey demand for RPS technology through 2020, the state could implement

economic incentives that would attract pertinent manufacturers and other related employers into New Jersey. If this occurred, the total annual economic impacts of the RPS from the installation and maintenance of both PV and off shore wind power would attain levels show in Table 2.15. That is, by 2020 a total of about 11,700 jobs would be created. In 2020 these jobs would annually support nearly \$1 billion in gross state product for the state, which would be composed, in part, by about \$700 million in job earnings and \$77 million in state and local government tax revenues in year 2000 dollars.

But naturally manufacturers would be unlikely to produce for the New Jersey market only. Indeed, data from R/Econ’s database report that New Jersey industries currently producing technology akin to that to be used to meet the RPS send about 70% of their product out of state and, on average, New Jersey manufacturers tend to ship about 80% of their production beyond New Jersey’s borders. Given that the RPS technology manufacturers would come to the state mostly to produce for that market, we assumed a more conservative level of out-of-state shipping—50%— for RPS technology manufacturers. Applying this assumption, we doubled the production from manufacturing industries to which we had assigned

Table 2.14 R/Econ I-O™ Annual Total Economic and Tax Impacts of the Installation and Maintenance of PV Panels, 2010 and 2020 (2000 Dollars)

	2010	2020
Jobs	520	2,600
Earnings	\$28,600,000	\$141,700,000
State & local taxes	\$2,600,000	\$12,900,000
Gross state product	\$38,700,000	\$191,400,000

**Table 2.15 R/Econ I-O™ Annual Total Economic and Tax Impacts of the Installation and Maintenance of RPS Power Generation Facilities
All Manufacturing of RPS Technology in New Jersey 2010 and 2020 (2000 Dollars)**

	2010	2020
Jobs	2,700	11,700
Earnings	\$159,200,000	\$694,100,000
State & local taxes	\$17,800,000	\$77,000,000
Gross state product	\$218,800,000	\$956,400,000

**Table 2.16 R/Econ I-O™ Annual Total Economic and Tax Impacts of the Installation and Maintenance of RPS Power Generation Facilities
50% of NJ Manufacturers of RPS Technology Production to NJ 2010 and 2020 (2000 Dollars)**

	2010	2020
Jobs	4,800	20,750
Earnings	\$267,000,000	\$1,139,000,000
State & local taxes	\$27,000,000	\$111,300,000
Gross state product	\$364,000,000	\$1,567,000,000

purchases to obtain results displayed in Table 2.15. The consequent economic impacts of this scenario are displayed in Table 2.16. Compared to the findings from Table 2.15, those in Table 2.16 show the number of jobs created by 2020 would increase by nearly 80% to 20,750 jobs and the annual gross product associated with those jobs would increase by a bit more than 60% to \$1.6 billion [14]. Annual earnings and state and local taxes revenues would be about \$1.1 billion and \$111 million, respectively. These results reflect the probable total economic impacts of a 20% RPS if New Jersey was able to be the regional center for PV and offshore wind manufacturing, installation and maintenance jobs.

III. Other Potential Impacts of a 20% RPS

The proposed 20% RPS has several other impacts not directly connected to macroeconomic issues. This section discusses the incremental reduction in natural gas prices and the reliability benefits, including possible reductions in transmission

[14] This order of increase is explained by several factors. First, manufacturing represents about 75% of the value of goods and services provided by industries during the installation, manufacturing, and maintenance of RPS technology. Second, the assumption that manufacturers ship 50% of their production out of state means that their production levels must be twice that needed to meet New Jersey's RPS demand. These first two factors would have us expect about a 75% increase in the economic measures. The third factor explains the variations around this expectation. The third factor is that manufacturing industries tend to pay workers higher than most others major sectors of the economy, particularly construction sector which comprises most of the rest of the spending for RPS technology installation and maintenance. The fact that manufacturing workers are paid more enables them to support more workers in the retail and personal service sectors, which are paid below the New Jersey worker average. Thus manufacturing jobs tend to support more low-paying jobs.

and distribution costs, attributable to the proposed 20% RPS.

The proposed 20% RPS would reduce the incremental demand for natural gas and therefore provide downward pressure on the price of this commodity. Several studies have quantified the relationship between reduction in natural gas demand and associated price reduction [Wiser et al.]. Based on these studies, each 1% reduction in national demand of natural gas leads to a long-term reduction in average natural gas wellhead prices of 0.75% to 2.5%, and some studies predict even larger reductions [Wiser et al.].

A 20% RPS would also contribute to distribution system reliability. The additional PVs would provide a source of power independent of the grid and local distribution system, assuming that these systems are designed to operate without power from the grid. PVs, however, are a small portion, 3.9% of the total renewables in 2020 under the proposed 20% RPS. PVs, if strategically placed, may avoid some transmission and distribution costs.

Regarding the reliability of the bulk power system, a 20% RPS may not provide any reliability benefits. The two major components of grid reliability are adequacy and security. The PJM capacity market is intended to ensure adequacy. Determining a resource's contribution to adequacy depends on its availability, which for PVs and wind is more volatile than traditional generation due to the random nature of sunlight and wind availability. If PJM appropriately accounts for the availability of PVs and wind, then its capacity requirement would reflect the availability of wind and PV and be sufficient to satisfy PJM's adequacy requirements. Security is the ability of the

grid to continue operating without interruption of service after the failure of generation or transmission elements. Failure of the largest system components are of primary concern. Renewable generation facilities, which are small compared to large power plants and major transmission components, are not likely to provide significant security benefits or concerns.

Having a substantial amount of wind power may pose additional costs related to reliability. According to a recent survey of studies on the reliability impact of wind on the bulk power system, the reviewed studies found that wind integration costs are positive and become more significant as wind power gains a greater share of electricity production, although these costs are small on a per-kWh of wind energy basis [Parsons, et al]. For wind penetrations of 5%, which would be approximately the amount of wind in the PJM region under the proposed 20% RPS, the additional cost are roughly \$2.00/MWh [Parsons, et al., Table 6] [15]. The Navigant Report, which provides the cost estimates for this report, assumes an additional operations and maintenance charge of \$4.00/MWh to account for various grid integration costs such as scheduling, regulation, and reserve requirements [Navigant, p. 23].

IV. Conclusion

The proposed 20% RPS increases the direct cost of electricity but has a negligible impact on the overall growth of the New Jersey economy. Expected technological advances that reduce the costs of PVs and wind are critical in order to minimize the electricity price impact and therefore the impact on the economy of the proposed 20% RPS. Higher fuel prices, however, reduce the electricity price impact of the proposed 20% RPS and at price levels assumed in the High Energy Price scenario, the proposed 20% RPS is economically more advantageous than the existing RPS.

The ability to locate PV and wind facilities, either from off shore facilities located in New Jersey or the bulk of manufacturing and support facilities for windmills located elsewhere, would contribute to the economic benefits of the proposed 20% RPS, although these benefits are small relative to the whole New Jersey economy. The proposed 20% RPS would also lower natural gas prices for New Jersey consumers, and PVs would provide backup power when the grid is not available.

The analysis conducted in this chapter does not include the environmental and other benefits of the proposed 20% RPS compared to the existing RPS. These additional benefits are discussed extensively in Chapter 3.

[15] The Navigant Report included interconnection costs of wind up to 0.5 miles from the grid.

Chapter 3: Environmental Benefits of a New Jersey Renewable Portfolio Standard

I. Introduction and Findings

A New Jersey RPS will reduce emissions generated by power plants that adversely affect the health and welfare of its population. Conceptually, the benefits from these reductions can be translated into dollar savings stemming from the reduction in mortality, morbidity, economic and other costs due to fewer emissions. This chapter traces the connection between emission reductions and their benefits and discusses approaches how to quantify these benefits.

Although it is possible to develop New Jersey specific estimates of the environmental benefits of its proposed RPS, the discussion in this chapter indicates that this would involve extensive data collection, modeling, and analysis, and is beyond the scope of this study. However, policymakers must make decisions in the interim in the absence of such data. Environmental externalities should not be ignored because of the difficulty in quantifying them. This chapter, therefore, also discusses the use of externality adders found in the literature and used by other policymakers as a substitute for developing New Jersey specific numbers. It also applies various externality adders to the New Jersey proposed RPS. Chapter 4 discusses the associated policy implications and recommendations of the proposed 20% RPS.

This chapter fits within an economic framework of externalities and this study's approach of identifying and quantifying the costs and benefits of New Jersey's proposed RPS. In general, externalities are costs or benefits that are not borne by individual producers or consumers and therefore are

external to their decision-making. In the presence of negative externalities such as emissions, market-based outcomes alone will result in more emissions being produced than socially optimal [16].

This chapter finds the following:

1. There are many and substantial health and environmental effects due to air emissions from power plants;
2. The health and environmental benefits from a RPS result from and depend on the reduction in the atmospheric concentration of emissions, which are (with the exception of carbon dioxide) geographic specific;
3. The health and environmental benefits of reduced emission concentrations are positive and span a wide range of values;
4. Quantifying the health and environmental benefits associated with a New Jersey RPS involves many detailed assumptions and using prior studies that may not be completely relevant to New Jersey. Therefore

[16] Although the approach of this report is to determine whether these non-market benefits can be monetized in the context of NJ RPS policymaking, there are other ways of accounting for these benefits in policymaking without monetization. One method is tradeoff analysis. If this approach were to be applied to this analysis, then each of the non-market benefits would be quantified separately and not combined into a dollar value. Tradeoffs between different impacts, such as costs and illnesses due to sulfur dioxide emissions, would be evaluated. This analysis can also be informed by stakeholders' values. Another approach involves using a deliberative process, for example one mandated by law, in which various stakeholders provide input.

a New Jersey analysis requires additional research and modeling in order to quantify these benefits;

5. Existing cap-and-trade emission allowance policies for sulfur dioxide and nitrogen oxides act in combination with a RPS so that the RPS may not alone result in reduced levels of these emissions but will lower the price of emission allowances;
6. Other policymakers use a wide range of externality values that may have limited application to New Jersey but these values are used in this report for illustrative purposes; and
7. Illustrative calculations using generic environmental externality adders indicate that in the year 2020 the environmental benefits of the proposed 20% RPS are in the range of several hundred million dollars.

Section II of this chapter describes the methodology used to quantify and monetize the health and environmental benefits of a New Jersey RPS. It consists of an overview of this methodology, a description of the New Jersey context, an analysis of the interaction between a RPS and emission allowance policies, and a summary of the literature of environmental externalities and electricity generation. This section also calculates the reductions in air emissions expected from a RPS and then uses the data presented to illustrate the calculation of the associated health and environmental benefits of air emission reductions.

Section III is a detailed review of the scientific and economic literature associated with quantifying the health and environmental effects of air emissions. It analyzes each emission, traces its health and environmental impact, and discusses how to quantify these impacts. Since the literature being reviewed is extensive, a detailed technical Appendix is also provided. Appendix B is a stand-alone document that also contains sample calculations that illustrate the types of calculations and their steps if New Jersey specific health and environmental effects from air emission reductions were to be conducted.

II. Monetizing the Benefits of Reducing Power Plant Air Emissions

Overview of the Methodology

Power plants have air, water, and solid waste emissions that adversely affect human health and the environment. The focus is on air emissions because the largest reductions in emissions due to a RPS are anticipated to occur in this category [17]. The air emissions that are of concern are carbon monoxide (CO), carbon dioxide (CO₂), nitrogen oxides (NO_x), particulate matter (PM), sulfur dioxide (SO₂), and mercury.

Figure 3.1 illustrates the steps this report follows to estimate non-market benefits of air pollution abatement: 1) identifying benefits, 2) quantifying benefits, and 3) monetizing benefits. The first step involves describing a qualitative relationship between changes in pollutant emissions and ambient concentrations, and subsequently between ambient concentrations and health and environmental effects. Identifying the benefits of air pollution abatement is equivalent to identifying the damages that are reduced or avoided. These damages fall into three broad categories (Freeman, 1993):

1. Direct damages to humans (e.g., increased asthmatic attacks);
2. Indirect damages to humans through ecosystems (e.g., reduced crop yields);
3. Indirect damages to humans through nonliving systems (e.g., damage to buildings).

Direct damages to humans include health damages, as well as aesthetic damages such as unpleasant odor, noise or poor visibility. Indirect damages to humans through ecosystems consist of productivity damages in the form of crop reduction, damages to forests and commercial fisheries, recreation damages (lakes, rivers, etc.), and intrinsic or

[17] Power plants may also emit solid wastes and liquid effluents that may contain toxic substances. According to one source [Bent et al], these are not generally an issue with existing environmental regulations. In addition, some power plants return water at different temperatures to rivers or other water bodies that harms plant and animal life. There are other possible environmental impacts as well.

nonuse damages. The latter are damages to ecological resources that are not motivated by people's own use of these resources. For example, people value endangered species or rare ecosystems, even though they do not have the intention to ever see or experience them. Finally, indirect damages to humans that occur through nonliving things include damages to materials and structures, such as soiling and corrosion.

The second step, quantifying benefits, involves establishing a functional relationship between environmental effects and the reduction in air pollution. Although an RPS policy reduces the amount of emissions, it is the reduction in concentration that results in the environmental benefits. Concentration-response functions quantify the relationship between the impact and emission concentration. For example, these functions describe the change in a health effect such as asthma attacks, and the concentration of the pollutant that causes the effect. In order to calculate the number of cases that will be avoided due to an RPS, a baseline exposure (number of people affected, and the level of pollution they are subjected) must be established and the baseline number of cases for each quantifiable health effect and for each pollutant. These numbers are then contrasted with the number of cases for each quantifiable health effect with the RPS regulation, to calculate the number of cases avoided as a result of the RPS.

The third step, monetizing or valuing the benefits, is typically specific to the environmental effect under consideration. Further below, we describe the most common valuation methods that have been used in the literature for each effect separately.

Understanding the benefits of reduced air emissions due to RPS policies requires tracing the emission pathways from the power plant to the damage site, illustrated in Figure 3.2. Starting at the top of this figure, a RPS policy reduces air emission by replacing existing sources of electricity with renewable resources. The reduced levels of emissions are transported through the atmosphere and deposited. This transportation process is affected by numerous conditions, such as the type of emission, weather conditions, the presence of other compounds in the atmosphere, etc. The result is a change in concentration of various emissions in the atmosphere. In most cases,

a reduction in emissions results in a reduction in concentration of the associated compound that results in damage. It is possible, however, to increase the level of ozone by reducing the concentration of NO_x , which combines with volatile organic compounds (VOCs) to form ozone. More details on specific emissions and their associated effects are discussed below.

As indicated in the middle of Figure 3.2, the benefits, i.e., avoided damages, of reducing air emissions from power plants result from the reduction of the concentration of air emissions not in the reduction in emissions. The benefits also depend on the level of concentration of the emission in the atmosphere as well as the amount of reduction from the initial concentration. As noted above, the relationship between the avoided damage and the change in concentration is the dose-response relationship. The damage due to emissions may be local, regional, or global.

Finally, as the bottom of Figure 3.2 indicates, to compare damages on a common scale, they need to be monetized. There are several ways to do so such as contingent valuation, travel cost, hedonic prices, market prices, and replacement costs. We discuss below each of these methods and their application to a New Jersey RPS policy. The sum of the monetized benefits, that is the costs associated with emissions that are avoided due to a RPS policy is the total environmental benefit.

Overview of Air Emissions in New Jersey

According to the most recent Air Quality Report published by the New Jersey Department of Environmental Protection (NJ DEP) [18]:

“Air Quality in New Jersey has significantly improved since the passage of the Clean Air Act in 1970....These improvements are the result of aggressive pollution control programs implemented in New Jersey as well as regional emission reduction strategies involving other states. But air quality problems do remain in the state.”

[18] Available at <http://www.state.nj.us/dep/airmon/02rpt.htm> as of September 23, 2004, Introduction, p. 1.

Figure 3.1: Taxonomy of Major Non-market Benefits of Air Pollution Abatement

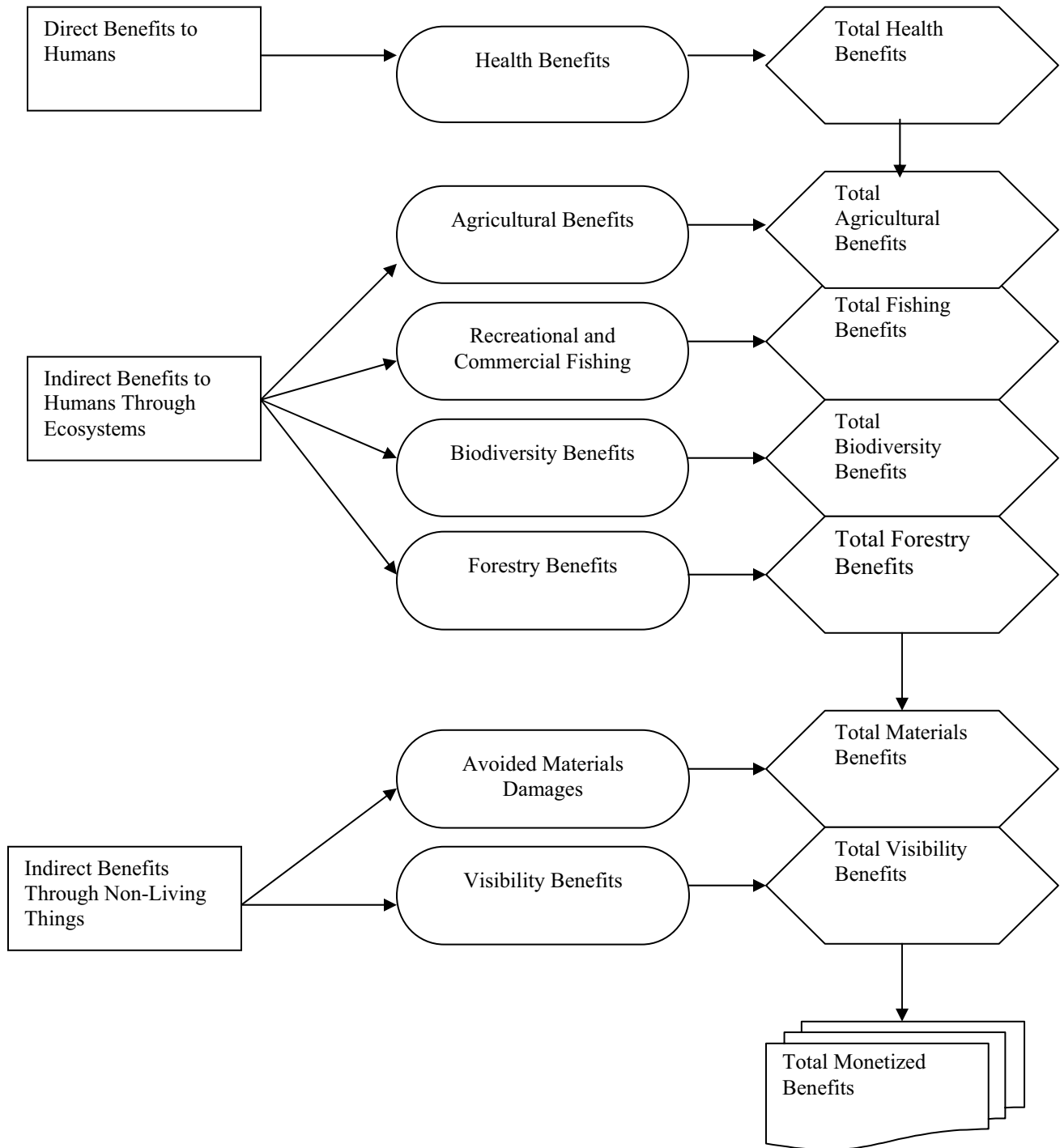
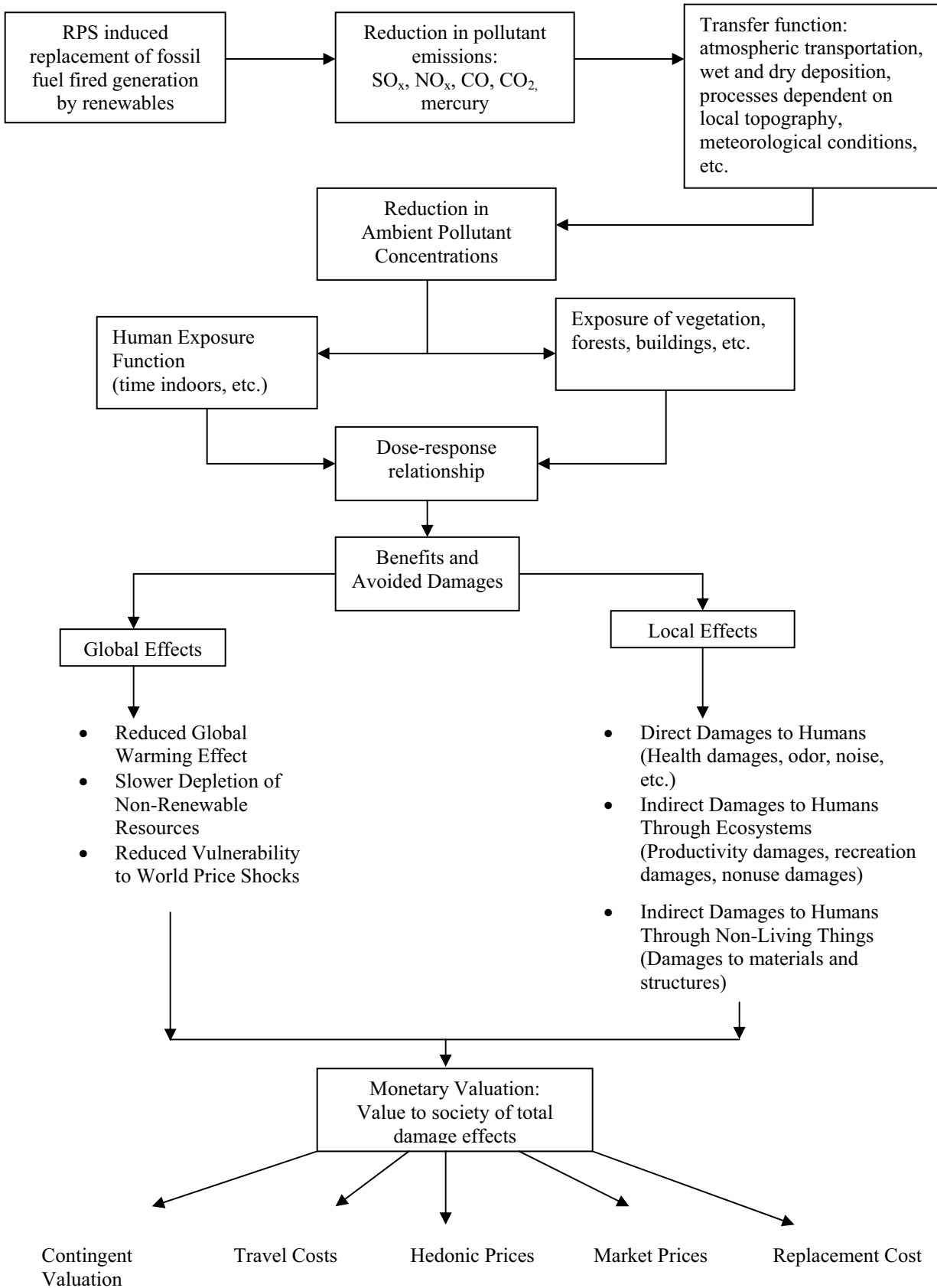


Figure 3.2: Links in the Environmental Chain Associated with RPS



In April 2004, the NJ DEP concluded that air pollution is a serious public health and environmental problem in the state, and highlighted three pollutants (ozone, fine particulates, and air toxics) that require immediate attention [19]. This section provides a high-level overview of air emissions from power plants, their pathways, their health and environmental effects, and the existing regulatory framework [20]. A complete discussion of these issues can be found in the New Jersey Department of Environmental Protection references cited below.

Air Toxics

Air pollutants can be divided into two categories: the criteria pollutants (ozone, sulfur dioxide, carbon monoxide, nitrogen dioxide, particulate matter, and lead) and air toxics. Air toxics are all other chemicals that can be released into the air and can cause adverse health effects in human (NJ DEP, 2002). In New Jersey, roughly 7% of air toxics come from major point sources, which include factories and power plants (NJ DEP 2004). Mercury is an air toxic that creates serious health problems. Over 2,000 pounds of mercury are emitted each year by sources in New Jersey, including its coal-fired power plants as well as municipal solid waste incinerators and scrap-melters (NJ DEP, 2004). Unlike criteria pollutants, for which the U.S. Environmental Protection Agency (EPA) has set National Ambient Air Quality Standards (NAAQS) and are subject to monitoring, reporting, and control requirements, there are no NAAQS for air toxics [21].

Carbon Dioxide (CO₂)

Carbon dioxide is produced as a by-product of the combustion process in coal, oil, and natural gas-fired power plants. CO₂ is a major greenhouse gas, which is widely believed to contribute to global warming. The amount of global warming is related to the concentration of CO₂ in the earth's atmosphere. Therefore, the location of the CO₂ emission and its

pathway, which is released immediately into the atmosphere, does not affect its contribution to global warming. Currently, there is no legal or regulatory restriction on the emission of CO₂ in the United States, although the governor of New Jersey has recently declared carbon dioxide a public hazard [22].

Carbon Monoxide (CO)

Carbon monoxide is a colorless, odorless and poisonous gas that is formed when carbon in fuels is not burned completely. It is a relatively unstable gas and its impacts are generally local to the source. About 82% of all CO emissions nationally are from the transportation sector, although CO is also produced in generating electricity from fossil fuels [23]. Nationwide, approximately 4% of CO comes from non-transportation, fuel-combustion sources such as boilers and incinerators (NJ DEP 2002). CO levels are typically higher in the winter because motor vehicles do not burn fuel as efficiently when they are cold and because atmospheric inversions that trap CO are also more frequent in the winter. CO exposure can cause headaches and nausea, is a threat to those that suffer from cardiovascular disease, and has other less severe effects such as reduced work capacity and decreased learning ability (NJ DEP, 2002).

Nitrogen Oxides (NO_x)

Nitrogen oxides are released as part of the combustion process in power plants, motor vehicles, and other sources of combustion. NO_x itself causes health and environmental damages but also combines with volatile organic compounds (VOCs) in the presence of sunlight to form ozone in the lower atmosphere. NO_x exposure can cause health effects such as respiratory problems and contribute to damage to ecosystems (NJ DEP, 2002). NO_x is regulated under emission allowance program administered by the U.S. Environmental Protection Agency [Ellerman et al]. The total amount of NO_x emissions is capped in the Northeast, and emitters buy emission allowances in order to emit NO_x or can sell any extra allowances to other emitters [24].

[19] NJ DEP, 2001, p. 1.

[20] See EIA, 1995 for a more complete treatment of the regulation of environmental externalities in U.S. electricity generation

[21] In 1990 the U.S. Congress directed that the EPA begin to list approximately 200 air toxics, known as Hazardous Air Pollutants (HAPs) (NJ DEP, 2004).

[22] The Record, Hackensack, NJ, September 17, 2004.

[23] NJ DEP, 2002, p. 1.

Ozone (O₃)

The formation of ground-level O₃ is a complex process requiring NO_x, VOCs, and sunlight. Man-made sources of VOCs include motor vehicles, chemical plants, factories, and consumer and commercial products. Ozone and its precursor pollutants can be transported hundreds of miles. According to the NJ DEP, much of the state's ozone comes from NO_x emitted upwind of the state from the Midwest and Southeast [25]. In 2002, the previous ozone standard was exceeded on 16 days and the new standard was exceeded on 44 days in New Jersey [26]. The NJ DEP concludes that attaining the new federal ozone standards in New Jersey would eliminate about 40,000 asthma attacks each year and substantially reduce hospital and emergency room admissions (NJ DEP, 2004).

Particulate Matter (PM)

Particulate air emissions refer to both solid particles and liquid droplets suspended in the atmosphere, and particulates can either be emitted directly or be formed from gaseous emissions, such as SO₂ and NO_x (NJ DEP, 2001) [27]. Particulate matter (PM) is categorized by its size. PM₁₀ refers to particulate smaller than 10 microns [28]. Particulates larger than this size are typically trapped in the respiratory tract prior to reaching the lungs. Fine particulate matter, PM_{2.5}, pose significant health impacts in New Jersey because these particulates can reach deep into the lungs (NJ DEP, 2004). Sources of fine particulates in New Jersey are diesel-powered engines and upwind power plants, whose gases are converted to PM as they travel downwind to New Jersey (NJ DEP, 2004). According to the NJ DEP, exposure to fine particulate levels above the federal health standards results in an estimated 350 to 1,200 deaths, 6,000 emergency room

visits, and 68,000 asthma attacks per year (NJ DEP, 2004).

Sulfur Dioxide (SO₂)

Sulfur dioxide is emitted as part of the combustion process involving fuels containing sulfur such as coal and oil. It, along with NO_x, has adverse impacts on terrestrial and aquatic ecosystems and damages buildings and other structures through the deposition of acid [29]. SO₂ can also lead to reduced visibility and cause irritation of the mucous membranes (NJ DEP, 2002). Sulfur dioxide can be transported long distances, such as from the Midwest to the Northeast before its deposition. Similar to NO_x, SO₂ is regulated under a cap-and-trade emission allowance program administered by the U.S. Environmental Protection Agency (Ellerman et al and Tietenberg). The net effect in the reduction of NO_x and SO₂ emissions between a RPS and emission allowance program are discussed below.

Interaction between Emission Allowance Programs and a Renewable Portfolio Standard

SO₂ and NO_x emissions are capped, and emitters must have sufficient allowances, which can be bought and sold in associated emission markets, to cover their actual emissions. Reductions in these emissions from a RPS frees up allowances that would have been used but for the RPS. These allowances then can be sold to entities that can use the allowance to emit the associated pollutant. If there were sufficient demand for allowances beyond the reduction due to the RPS, then a RPS would not lower the emission of SO₂ and NO_x but only the price of the allowances.

Figure 3.3 illustrates this situation in which an emission cap combined with a RPS does not result in a net reduction of emissions. It shows a vertical supply curve for the emissions because the emission

[24] An RPS with RECs (renewable environmental credits) is conceptually the same type of policy as an emission allowance trading policy, except that the emissions are a negative externality and renewables are a positive externality.

[25] NJ DEP, 2004, p. 4.

[26] NJ DEP, 2004, p. 4.

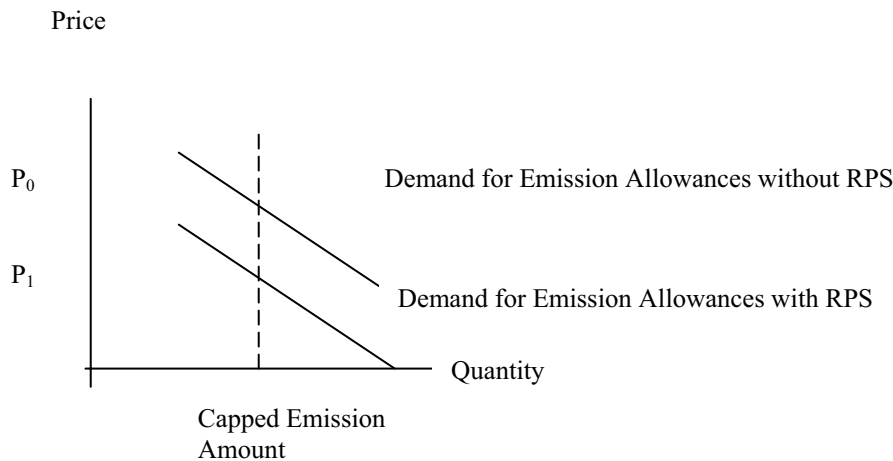
[27] The most recent publicly available air quality report from the NJ DEP on particulates is 2001.

[28] One micron is one millionth of one meter, also called micrometer, denoted by μm .

[29] Acid deposition can take two forms: wet deposition and dry deposition. Wet deposition occurs when sulfur dioxide and nitrogen oxides in the atmosphere react with water and return to the earth's surface as acidic water, commonly referred to as acid rain. During dry deposition sulfur dioxide and nitrogen oxides settle out of the atmosphere as particles or gases.

is capped and two downward sloping demand curves. The higher curve is the demand curve for emission allowances without a RPS, and the lower curve reflects the reduction in demand for allowances due to the RPS. As Figure 3 indicates, the price for the allowance decreases from P_0 to P_1 but the quantity of the emissions does not change.

Figure 3.3 Net Effects of Emission Caps and a Renewable Portfolio Standard



Because of the effect just described, the U.S. Energy Information Agency concluded that a national RPS is projected to have little impact on SO_2 and NO_x emission levels but would result in a significant reduction in SO_2 allowance prices (EIA, 2003) [30]. Of course, the reduction in allowance prices due to a RPS is a benefit of the policy, but this is an economic benefit not a health or environmental benefit due to reduced concentration. The impact, and therefore environmental benefit, can be captured if the RPS results in a reduction of the cap. While the NO_x cap is periodically reduced, it is not clear what precise role that the RPS policies have in that calculation. Accordingly, one recommendation of

[30] The EIA also found that there would be little reduction in NO_x allowance prices because the RPS that it analyzed permitted co-firing. Primarily a generation unit's boiler type and emission control type, rather than its fuel drives NO_x emissions. The Minnesota Public Utilities Commission made a similar finding with respect to SO_2 : "Regarding post-2000 issue, the Commission finds that the SO_2 damages will be internalized after 2000 and, therefore, applying externality costs would be unwarranted (MN PUC)." Minnesota is not part of a NO_x cap-and-trade allowance program.

this study, discussed more in Chapter 4, is that New Jersey aggressively seek and lead a coalition of states to obtain specific reductions in established caps to capture all of these environmental benefits.

Externality Adders

Review of Externality Adders

An extensive literature exists that attempts to quantify the environmental externalities from generating electricity. The goal of this literature is to provide externality adders, which vary by the type of fuel used, to reflect the health and environmental cost of generation beyond the cost of producing that electricity. Policymakers can then use these adders to calculate the environmental benefit of various policies. Adders are commonly reported in units of cents per kilowatt-hour (kWh) of electricity.

This section summarizes the results of this literature based on a recent comprehensive, international, and scholarly review (Sundqvist) [31]. The literature provides a wide range of methodologies and approaches to this problem, and the result is a broad array of estimates for externality adders.

[31] See also Office of Technology Assessment, NREL, and EIA 1995.

Table 3.1 presents these results. Note the wide range of estimates for natural gas externality adders. The maximum (13.22 cents/kWh) is approximately 400 times larger than the minimum estimate (0.003 cents/kWh); the mean and median differ by almost by a factor of two. Natural gas is highlighted because as discussed below, it is the marginal fuel for electricity generation, and it is the fuel that will be displaced by renewable resources.

As Section III reveals, externality adders depend upon numerous assumptions, the results of scientific studies that contain ranges of estimates for a particular parameter, and must account for many specific conditions that may not be transferable between the situation the policymaker is addressing and the availability of scientific evidence. For instance, air concentrations of a pollutant may be different in rural than in urban areas.

Table 3.2 lists the results of U.S. studies and has similar ranges of estimates as Table 3.1. Again note the wide range in the estimates for the natural gas externality adder. The minimum value reported in this table is 0.003 cents/kWh and the maximum is 7.98 cents/kWh, a range that spans three orders of magnitude.

States have taken a wide range of approaches in valuing externalities (EIA, 1995). Typically, values are used that are specific to emissions, e.g., \$/ton of SO₂ or

\$/ton of CO. Although dated, the Energy Information Agency provides a state-by-state summary of public utility commissions' (PUC) activities regarding externalities (EIA, 1995, Appendix). According to this state-by-state summary, Pennsylvania did not require the consideration of externalities and New Jersey gave a 2 cents/kWh (1991 \$) credit to demand-side management projects. Delaware required qualitative but not quantitative consideration of externalities. Maryland did not have an order that mandates the use of externalities but does consider externalities qualitatively and quantitatively. Table 3.3 compares New York and Massachusetts externality adders. States have not been active in this area for several years due to the transition to restructured wholesale electricity markets. It is also important to note that states typically determine externality adders as part of a negotiation process.

Minnesota provides a more recent example of a state's externality values than New York and Massachusetts. They are listed in Table 3.4.

Illustrative Calculation of the Environmental Benefits of the Proposed 20% RPS

To illustrate how to quantify the health and environmental benefits due to a 20% RPS, various estimates of environmental adders are applied.

Table 3.1 Summary of Externality Adders from Power Generation Externality Studies

Cents/kWh (1998 \$)	Coal	Oil	Gas	Nuclear	Hydro	Wind	Solar	Biomass
Minimum	0.004	0.03	0.003	0.0003	0	0	0	0
Maximum	67.72	39.93	13.22	64.45	26.26	0.88	2.2	22.09
% Difference	16930%	1331%	44100 %	214833%	--	--	--	--
Mean	14.01	12.32	4.61	7.12	3.36	0.31	0.84	4.95
Median	6.38	9.11	2.62	0.81	0.32	0.32	0.76	2.68
Standard Deviation	15.99	12.45	4.58	16.96	7.59	0.24	0.74	5.57
Number of Studies	36	20	31	21	16	18	11	22

Source: Sundqvist, Table 3

Table 3.2 Summary of Externality Adders from Power Generation Externality Studies Conducted in the United States

Cents/kWh (1998 \$)	Coal	Oil	Gas	Nuclear	Hydro	Wind	Solar	Biomass
Schuman & Cavanagh (1992)	0.06-44.07	--	--	0.11-64.45	--	0.025	0-0.25	--
Chernick & Cavehill (1989)	4.37-7.74	4.87-7.86	1.75-2.62	--	--	--	--	--
Bemow & Marron (1990); Bemow et al. (1991)	5.57-12.45	4.40-12.89	2.10-7.98	--	--	--	--	--
Hall (1990)	--	--	--	2.37-3.37	--	--	--	--
Ottinger et al. (1991)	3.62-8.86	3.87-10.36	1.00-1.62	3.81	1.43-1.62	0-0.12	0-0.5	0-0.87
Putta (1991)	1.75	--	--	--	--	--	--	--
Cifuentes & Lave; Parfomak (1997)	2.17-20.67	--	0.03-0.04	--	--	--	--	--
ORNL & RfF (1994-1998)	0.11-0.48	0.04-0.32	0.01-0.03	0.02-0.12	0.02	--	--	--
RER (1994)	--	0.03-5.81	0.003-0.48	--	--	--	--	--
Row et al. (1995)	0.31	0.73	0.22	0.01	--	0.001	--	--

Table 3.3 Comparison of New York and Massachusetts Externality Adders

Pollutant	Massachusetts Externality Adders (2003 \$ per ton)	New York Externality Adders (2003 \$ per ton)
NO _x	\$9,620/ton	\$8,546/ton
SO ₂	\$2,220/ton	\$1,791/ton
VOC	\$7,844/ton	\$5,764/ton
Total Suspended Particulates	\$5,920/ton	no value assigned
CO	\$1,288/ton	\$1,130/ton
CO ₂	\$33/ton	\$554/ton
CH ₄ (methane)	\$326/ton	no value assigned
N ₂ O	\$5,861/ton	no value assigned
CFCs	no value assigned	no value assigned
Air Toxics	no value assigned	\$214,840/ton
Water use, land use, ash disposal	no value assigned	no value assigned
Note: Values were calculated using CPI data from the Bureau of Labor Statistics		

Source: (EIA, 1995, pp. 34 and 86)

Table 3.4 Final Environmental Cost Table for the State of Minnesota (ME3)

Emission (2002 \$/ton)	Urban	Metropolitan Fringe	Rural	Within 200 Miles of Minnesota
SO ₂ (after year 2000 - \$/ton)*	0	0	0	0
PM10(\$/ton)	5,060 - 7,284	2,253 - 3,273	637 - 970	637 - 970
CO(\$/ton)	1.20 - 2.57	0.86 - 1.52	0.24 - 0.46	0.24 - 0.46
NO _x (\$/ton)	421 - 1,109	159 - 302	20 - 116	20 - 116
Pb(\$/ton)	3,551 - 4,394	1,873- 2,262	456 - 508	456 - 508
CO2(\$/ton)	.34 - 3.52	.34 - 3.52	.34 - 3.52	0

* The zero values for SO₂ are explained in the discussion on the interaction between an RPS and cap-and-trade emission allowance program.

The marginal fuel in the PJM market, as well as throughout the United States, is natural gas. In 2005, the percentage of megaWatt-hours fueled by natural gas in PJM is expected to be 38.7% and decrease slightly to 34% in 2020 [EIA, 2004 Supplemental, Table 62]. According to a list of proposed new generation power plants in PJM covering the next five years, the majority of new power plants in the region and in New Jersey are fired by natural gas. More important than the current and future percentage of natural gas in the electricity fuel mix is the relative variable cost of producing electricity from natural gas compared to other traditional technologies. Once a power plant is built, its variable costs determine how often a unit is dispatched. Nuclear and coal power plants have very low variable costs compared to natural gas-fired plants. For instance, the variable costs of an advanced coal plant are about a third of those of an advanced combined cycle, and even less for a gas-fired turbine (EIA Outlook 2004). Renewable resources will displace high variable cost power plants, that is, ones fueled by natural gas [32].

Table 3.5 calculates the incremental environmental benefits of avoiding generating natural gas due of the proposed 20% RPS. It uses the externality adder of \$0.0216/kWh (in 2004\$), which is the difference between the median externality adder of natural gas fired generation and the median externality adder for PV presented previously. (The difference between the natural gas and wind externality adders

is \$0.0267/kWh (in 2004\$), which is even larger.) By using the median externality values and the higher PV value, the estimated benefits are conservative values. Table 3.5 does, however, assume that the incremental reductions in SO₂ and NO_x due to the proposed 20% RPS are captured by New Jersey.

Table 3.5 Illustration of Incremental Environmental Benefits of a 20% RPS (in 2004 million \$)

	2010	2015	2020
Environmental Benefit	\$ 12	\$ 110	\$ 330

The above calculations are based on non-state specific estimates of the benefits of avoiding natural gas fired generation not on a specific New Jersey analysis. The next section reviews in detail the health and environmental effects of air emissions and the methodology to quantify in dollars their impact. Section III provides a template for determining New Jersey specific externality adders.

III. Detailed Review of Health and Environmental Effects of Air Emissions

In this section, a detailed literature review is discussed. The major findings of this section is that New Jersey specific health and environmental benefits of reducing air emissions from a RPS could be determined using a benefit transfer approach in conjunction with modeling the New Jersey atmosphere (air shed modeling). In addition, existing concentration levels of the pollutants of interest across New Jersey spatially and temporally are needed along with the number and duration of exposure. Applying

[32] The fact that natural gas-fired units are operating in substantial amounts for most hours of the year was confirmed by running a unit commitment and dispatch model of PJM in selected years between 2010 and 2020.

any existing air shed modeling or conducting it is, however, beyond the scope of this report. In Chapter 4, we discuss whether such an effort should be undertaken for New Jersey. This literature review also demonstrates the types of assumptions necessary for these types of calculations and provides some insight into why there are substantial ranges for the health and environmental impact of air emissions.

Benefit Transfer Approach

Local conditions affect the impact of emissions, such as the existing concentration of the emission, susceptibility of the population and environment to the type of damage caused by the emission, etc. Since conducting detailed and high quality studies for each emission and type of damage is expensive and time consuming for every possible local condition, researchers have developed an alternative, known as a benefit transfer approach.

In a benefit transfer approach, the results of other studies conducted in other regions of the country are reviewed to determine their applicability to the region of interest. Ideally, primary research based on the characteristics of New Jersey is preferred to a benefit transfer approach to developing the dose-response curve, but the types of benefits are too numerous and their phenomenon too complex to pursue this approach given the constraints of this specific project. The validity of this benefit transfer approach depends on how closely the other studies resemble the situation in New Jersey based on timeliness, methodology, and other factors. In addition, as part of the process of evaluating the evidence presented by this body of literature, each study must be evaluated as to the soundness of the data and the analysis techniques and the conclusions drawn by the authors.

A benefit transfer approach is applicable when existing studies value similar effects, when the context of the existing studies is highly similar, and when the existing studies are of high quality. Benefit transfer may increase the inherent uncertainty surrounding environmental benefits estimates, but for practical reasons (e.g. cost and time), benefit transfer is a necessary component of policy analysis. In most situations, the question is not whether to conduct benefit transfer, but how to improve benefit transfer to

make it more reliable.

There are no universally accepted criteria for benefit transfer, but in most cases the defensibility of the benefit estimates depends on the quality of the existing study. We use the following two criteria. First, we selected studies that were published in peer-reviewed journals and cited in the academic literature. In addition, the sources of variation have to be considered between the original study location/situation and the one to which the benefits estimates are transferred. We focus on studies conducted in North America. We used the 1999 U.S. Environmental Protection Agency report (US EPA, 1999) as a baseline and conducted a thorough literature review of studies completed after that report. We cite all studies that met our criteria either below or in our technical Appendix.

Direct Benefits to Humans – Health Benefits

Identifying Health Benefits

Over the last 15 years, a large body of epidemiological literature has been devoted to the study of adverse health effects occurring at moderate and low pollutant concentrations. There is now sufficient evidence to support the hypothesis that both chronic and acute health effects can occur at ambient pollution levels. Current research focuses on the consequences of acute and chronic air pollution exposure for excess cardiovascular and respiratory morbidity and mortality. (Appendix, Section 4.)

Health benefits resulting from reduced air pollution can be grouped into two broad categories: mortality benefits and morbidity benefits. Mortality benefits are avoided deaths from diseases caused by air pollution. Morbidity benefits refer to avoided cases of non-fatal health effects. Air pollutants that have been linked to adverse health effects include particulate matter (PM), ozone (O₃), carbon monoxide (CO), sulfur dioxide (SO₂), and nitrogen dioxide (NO₂). Until the mid-1980's it was believed that ambient pollutant concentrations did not have adverse health effects [33].

[33] Katsouyanni (2003)

Types of Health Studies

There are two main types of methods to establish a dose-response relationship for a health effect: epidemiological studies and toxicological studies. Our literature review surveys both types.

Epidemiological studies attempt to establish a quantitative relationship between health effect and air pollution using a cross-sectional sample drawn from a large population. The four most commonly used types of epidemiological studies are cohort studies, case control studies, occupational epidemiology studies, and cross-sectional studies.

In toxicology, mechanistic studies examine how and why various disease processes occur in response to toxicant exposures, and help establish a relationship between dose or exposure and response at the molecular level. For example, an animal study provides direct information about the subject's adverse response to a substance and its level. One of the advantages of animal studies is the researcher's ability to extrapolate from the high doses in animal studies down to the low doses often experienced in human exposure scenarios. Finally, human studies can be used to extrapolate the response of humans at low doses to higher doses.

Health Benefits Due To Particulate Matter (PM) Reductions

Adverse health effects of exposure to particles have been described in numerous epidemiological studies. Health impacts that are measurable and experienced by humans are referred to as *health endpoints* and include all-cause and cause-specific mortality and morbidity. Studies conducted in the United States and in other countries have reported associations between changes in PM and changes in mortality and morbidity, particularly among subgroups of people with respiratory or cardiovascular diseases. However, the exact mechanisms by which PM influences human health are not well understood. Earlier literature focused on PM greater than 10 microns in diameter, while in the last decade the attention of researchers turned to fine particles such as PM_{2.5}. Recent research indicates that ultrafine particles (UF) less than 0.1 micron in diameter may play an important role in the induction of toxic effects.

Currently, however, data on UF exposure and health effects are still limited.

Although the importance of long-term exposure to PM has been emphasized, most of the attention in the literature has been devoted to short-term health effects. Two prominent prospective-cohort studies of mortality effects of PM are Dockery et al. (1993) and Pope et al. (1995). Unlike earlier studies, Dockery et al. (1993) estimate the effect of air pollution on mortality, while controlling for individual risk factors. Pope et al. (1995) study the association between air pollution and mortality using data from a large cohort drawn from many study areas. Pope et al. (2002) is a continuation of the Pope et al. (1995), while the HEI (2000) study is a reanalysis of the original Pope et al. (1995) data. Fine particle and sulfur oxide pollution were associated with all-cause death, lung cancer and cardiopulmonary mortality. Each 10µg/m³ increase in fine PM pollution was associated with approximately 4%, 6%, and 8% increase in the risk of all-cause death, cardiopulmonary mortality, and lung cancer mortality, respectively (Pope et al, 2002). Information on data, methodology, and the results of these four studies are summarized in Appendix B.

Short-term or acute effects of PM are well established for morbidity endpoints such as, hospital admissions for respiratory and cardiovascular conditions. There is also evidence of acute effects on respiratory function, lower respiratory symptoms, and increased medication use by asthmatics [34]. There are fewer studies available on the long-term, or chronic, health effects of PM pollution. A few studies have linked an increase in chronic bronchitis occurrence to an increase in ambient PM concentration [35]. The studies reviewed find a positive association between the health effect and PM air pollution. Appendix, Section 4.2 summarizes the available studies that assessed morbidity effects resulting in chronic and minor illness, as well as hospital admissions.

Health Benefits Due To Ozone Reductions

Ozone is formed by a chemical reaction from its precursor pollutants (volatile organic compounds

[34] For additional references, see Katsouyanni (2003)

[35] Abbey et al. (1993), Schwartz (1993), Abbey et al. (1995)

(VOCs) and nitrogen oxides (NO, NO₂), and nitrous oxide (N₂O)) in the presence of heat and sunlight. Ozone concentration is the highest in the summer when the weather is hot and sunny with relatively light winds.

Health problems are caused by tropospheric, or ground-level, ozone. Ozone is associated with a variety of adverse health effects ranging from minor symptoms to hospital admissions and chronic illness. Some studies have found a link between ozone and mortality, however there is significant uncertainty about the relationship between mortality and high ozone concentrations, partly because of the possible confounding effect of other pollutants such as particulate matter [36]. Table 3.6 below summarizes the most common adverse health effects associated with ozone.

Mechanistic studies of ozone yield a sufficient evidence for a biologic plausibility of respiratory-

related morbidity and mortality [37]. There is evidence from human and animal exposure studied that long-term exposure to ozone may cause a sustained decrement in lung function. There are well-documented molecular mechanisms for acute respiratory effects of ozone, but the evidence for chronic respiratory effects is limited. There is, however, increasing evidence that high levels of ozone can result in the development of chronic diseases. For example, McConnell (2002) provided the first evidence suggesting that tropospheric ozone causes the development of childhood asthma. In high ozone-concentration cities, children who played outdoor sports were 3 to 4 times more likely to develop chronic asthma than children who did not play sports. In low ozone-concentration cities, children playing sports were no more likely to develop asthma than children who did not play sports.

Because indoor ozone concentrations are generally lower than ambient concentrations, personal exposure may not be directly related to ambient

Table 3.6 Likely Ozone-related Adverse Health Effects

Adverse Health Effect	Comment
Respiratory Hospital Admissions	Numerous studies have linked ozone to hospital admissions for pneumonia, chronic obstructive pulmonary disease (COPD), asthma and other respiratory ailments.
Cardiovascular Hospital Admissions	There is a link between high ozone and dysrhythmias (abnormal heartbeat patterns).
Total Respiratory ER Visits	Studies have also found a link between high ozone and emergency room visits which do not result in actual hospital admissions.
Minor Symptoms	Short-term exposure to ozone has been linked to a variety of symptoms, including cough, sore throat and head cold.
Asthma Attacks	Ozone has specifically been linked to incidence of asthma attacks and may be linked to the development of chronic asthma.
Shortness of breath	Ozone associated with shortness of breath in asthmatics and non-asthmatics.

Source: Abt Associates, Adverse Health Effects Associated with Ozone in the Eastern United States (Washington, D.C: Clean Air Task Force, October 1999).

[36] See for, example, Kinney et al. (1995), and Ito and Thurston (1996)

[37] For a review see of mechanistic studies of ozone, see Levy et al. (2001)

concentration. Personal exposures to ozone are influenced by air conditioning or averting behavior, such as, more time spent indoors. When applying concentration-response functions, the relationship between ambient ozone concentrations and personal exposures must be determined. An understanding of any systemic differences between the study and policy region is crucial. For example, if there are differences between indoor air quality or the type of averting behavior in New Jersey from that of studies conducted in other parts of the country used to estimate the health effects of outdoor ozone exposure, then the results may not be transferable. This example illustrates a limitation of the benefit transfer approach.

Health Benefits Due To Carbon Monoxide Reductions

Carbon monoxide (CO) is a colorless and odorless gas produced through incomplete combustion of carbon-based fuels. Carbon monoxide enters the bloodstream through the lungs and reduces the delivery of oxygen to the body's organs and tissues. The most vulnerable to CO are those who suffer from cardiovascular disease, particularly those with angina or peripheral vascular disease. Fetuses and young infants, children, pregnant women, individuals with obstructive pulmonary disease, such as bronchitis and emphysema, smokers, and individuals spending a lot of time on the street working or doing exercise are also more susceptible to CO exposure. In health studies, high CO concentrations have been linked to hospital admissions for asthma, chronic obstructive pulmonary disease (COPD), dysrhythmias, ischemic heart disease, and congestive heart failure (CHF) [38].

Health Benefits Due To Sulfur Dioxide Reductions

Sulfur dioxide is formed when fuel, containing sulfur, such as coal and oil, is burned. The main health effects associated with exposure to high SO₂ concentrations include effects on breathing, respiratory illness, changes in pulmonary defenses, and aggravation of cardiovascular disease. The most susceptible groups are children, the elderly, asthmatics,

and people with cardiovascular and chronic lung disease (such as bronchitis and emphysema). High SO₂ levels have been linked to the following endpoints: hospital admissions for pneumonia, ischemic heart disease, and respiratory conditions, chest tightness, shortness of breath, and wheeze [39].

Health Benefits Due To Nitrogen Dioxide Reductions

NO₂ is a suffocating, brownish gas that is formed when fuel is burned at high temperatures. Primary sources of NO₂ are motor vehicles, electric utilities and industrial boilers. Nitrogen dioxide can irritate the lungs and lower resistance to respiratory infections such as influenza. There is no clear evidence on the effect of short-term exposure to NO₂ on health, but frequent exposure may cause an increased incidence of acute respiratory illness, especially in children. NO₂ has been linked to hospital admissions for respiratory conditions, pneumonia, congestive heart failure, and ischemic heart disease. Epidemiological studies found that NO₂ has a modifying effect on PM: the increase in mortality due to PM was found to be higher in cities where long-term NO₂ concentrations were higher. In addition NO₂ may have other indirect adverse effects, as it contributes to ozone formation. Therefore the importance of NO₂ for health comes from its role as an O₃ precursor and a contributor to the formation of secondary particles [40].

Quantifying Health Benefits

Health benefits are typically estimated using the damage-function (DF) method that consists of the following:

1. Determining the dose-response relationship for each health effect
2. Determining baseline exposure
3. Determining the number of baseline cases for each quantifiable health effect

$$\text{Number exposed} \times \text{Baseline exposure} \times \text{Dose response relationship}$$
4. Determining exposure after the regulation (for each regulatory option)

[38] Burnett et al. (1999), Koken et al. (2003), Linn et al. (2000), Lin et al. (2003), Moolgavkar (1997), Sheppard (1999), Schwartz (1999), Schwartz (1999), Schwartz and Morris (1995)

[39] See Burnet et al, 1997b, Moolgvaka, 1997, and Lim et al., 1990.

[40] See Burnett et al, 1999, Burnett et al 1997b, Moolgvakar, 1997, and Katsiouyanni, 2003.

5. Determining the number of cases for each quantifiable effect with the regulation
6. Determining the number of cases avoided as a result of each regulatory option

The purpose of quantification is to determine the change in the occurrence of a health effect as a result of a change in pollutant concentrations between the baseline and the control scenario. Such a relationship between, say particulate matter (PM) concentration, and the change in the health effect is described by dose-response or concentration-response (CR) functions. Dose-response or concentration-response (CR) functions estimate the risk (of the occurrence of a health effect) per unit of exposure to a pollutant.

Relative risk (RR) is a measure that denotes the seriousness of exposure to a known risk factor. It is defined as the risk for those exposed relative to those who are not exposed. Exposure increases the risk because in a sufficiently large population, some people develop a disease that can be attributed to air pollution regardless of whether they are exposed or not. For example, if the risk of developing a disease in the exposed population is 5% while in the non-exposed population it is 1%, the relative risk is 5. A high relative risk indicates strong evidence between exposure to a pollutant and the health effect.

Typically, dose-response functions that have been estimated for health effects describe a linear no-threshold relationship. This means that every unit of exposure contributes equally to aggregate risk in a large population of people. For example, a linear no-threshold dose-response function treats the case of one person being exposed to one hundred units of the pollutant, and ten people subjected to ten units of pollution equally (simply, as 100 units of human exposure). Thresholds may be incorporated into the analysis even when one uses CR-functions that were derived under the no-threshold assumptions. While the possible existence of a threshold in concentration-response relationship is an important scientific question, there is currently no scientific basis for selecting appropriate threshold levels. A more detailed discussion of this issue is located in Appendix, Section 4.2.

The threshold versus no-threshold issue has an important policy implication for New Jersey. In 2002 all New Jersey air emission levels except for ozone were below the health standard (NJDEP, 2002). If there was a threshold, reductions in emission concentrations beneath that threshold, assuming that the safety standard is the threshold, would not provide any public health benefits whereas if there is not a threshold, even reductions beneath the safety standard would provide health benefits.

There are some important limitations in quantifying health effects of emissions. To determine the number of baseline cases, the following must be identified: the segment of population that is exposed, the number of people exposed within each segment, and the level, duration and frequency of exposure. In order to obtain accurate estimates, averting behavior (that is, people with known risk may act to avoid exposure) must be controlled.

Quantifying Mortality Benefits

In valuation studies, mortality benefits linked to particulate matter (PM) tend to dominate total monetized benefits of air pollution abatement [41]. The relationship between mortality and ambient PM concentration is well established, while this is not the case for other pollutants [42]. There is some evidence that there are synergistic effects between PM and other pollutants (e.g. ozone). For example, reduction of PM concentration may reduce the health impacts of exposure to other types of emissions such as ozone. Studies that control for these synergistic effects are preferable to ones that do not, and the studies that we reviewed typically accounted for these effects.

Quantifying Mortality due to Particulate Matter

In epidemiological studies of PM, typical measurable health endpoints include all-cause and cause specific mortality, as well as hospital admissions and emergency room visits. The Pope et al. (2002) study estimates relative risk associated with a 10 $\mu\text{g}/\text{m}^3$ increase in $\text{PM}_{2.5}$ (particulates less than 2.5 μm in diameter), while the HEI (Health Effects Institute) (2000) study consider a 25 $\mu\text{g}/\text{m}^3$ change. In the

[41] See for example, Stieb et al. (2002)

[42] USEPA (1999)

Pope et al. study, the relative risk from all causes of mortality from a 10 $\mu\text{g}/\text{m}^3$ increase in $\text{PM}_{2.5}$ from the years 1999-2000 is 1.06. In Appendix B, Tables 3.13-3.45, we list dose-response functions from available studies.

Quantifying Mortality due to Ozone

There is considerable epidemiological evidence concerning the relationship between ambient ozone concentrations and human mortality risks. Because ozone contributes to acute (short-term) health effects, the association between daily ozone concentrations and daily mortality is of primary interest to researchers. These studies show mixed findings as to whether there is a statistically significant association between daily ozone concentrations and daily mortality in each of the study areas. Depending on the study, the relative risk for a 25 ppb increase in ozone ranges from 1.00 to 1.06. See the Appendix for further details.

Quantifying Mortality due to Carbon Monoxide

A number of studies examined the relationship between daily mortality and concentrations of CO. Cardiovascular mortality was found strongly associated with CO concentrations. The Appendix summarizes the results and characteristics of the studies identified in our literature search. The relative risk for a 10 ppm increase in CO reported in the literature we reviewed ranges from 1.06 to 2.44.

Quantifying Morbidity Benefits due to Carbon Monoxide

Quantifying morbidity benefits is more difficult for chronic (long-term) conditions than for acute (short-term) health effects, because it requires data on exposure over a long period of time. The most frequently used endpoints in the epidemiological literature are hospital admissions for various respiratory and cardiovascular illnesses, emergency department visits, chronic diseases (e.g. chronic bronchitis), and minor health effects, such as upper respiratory symptoms (URS), lower respiratory symptoms (LRS), asthma attacks, shortness of breath, work loss days, minor restricted activity days, etc. In the Appendix, Section 4.2, additional studies are referenced regarding minor illness hospital admissions for respiratory and cardiovascular causes.

Valuation of Health Benefits

Monetizing Mortality Benefits

Environmental economics developed a number of methods for estimating health benefits from avoided air pollution. The most popular primary methods are described below:

- Contingent valuation method (*CVM*) is a survey-based method to determine willingness-to-pay (WTP) for a hypothetical change in environmental effects.
- Averting behavior is a method to infer WTP from the costs and effectiveness of actions taken to avoid a negative environmental effect.
- Cost-of-illness (COI) or damage costs methods involve estimating direct costs (such as, medical expenses) and indirect costs (for example, forgone earnings) of an environmental effect.
- Hedonic methods estimate WTP for an environmental amenity by inferring its value from the market price of another (but in some sense related) good. For example, the *hedonic property values* method estimates a marginal WTP based on an estimated relationship between housing prices and housing attributes (that include environmental amenities, such as good visibility or air quality). In contrast, the *hedonic wages* method estimates the value of environmental amenities from a worker's willingness-to-avoid (WTA) a higher salary to compensate for exposure to higher levels of risk on the job.

Mortality benefits are most often monetized using a Value of Statistical Life (VSL) estimate. VSL is a measure of the WTP for reductions in the risk of premature death aggregated over the population experiencing the potential risk reduction. For example, if each person out of one million people is willing to pay five dollars for a 1:1,000,000 reduction in mortality risk and on average one life is saved, then the value of VSL is \$5,000,000. (EPA relies on a composite VSL estimate based on 26 VSL studies – 21 of which use the hedonic wage method, and 5 use

CVM, EPA's current guidelines advise analysts to use a VSL estimate of \$6.2 million in 2002 dollars).

Despite its widespread use, the VSL concept has been subject to criticism. One shortcoming of the VSL is highlighted when considering the difference between mortality due to acute (short-term) and mortality due to chronic (long-term) exposure. One of the basic assumptions underlying the VSL approach is that that equal increments in fatality risk are valued equally irrespective of the initial risk. This assumption is defensible only if the prior risk is small. Some experts have suggested that it is not appropriate to estimate mortality that is the result of acute exposure, because people affected usually have a pre-existing disease and a relatively high prior risk of mortality [43]. The primary approach of estimating VSL has been the use of hedonic wage and hedonic price models that examine the equilibrium risk choices. The observed market decisions (measured by wages and prices) reflect the joint influence of supply and demand in the market. Most of the empirical literature has concentrated on valuing mortality risk by estimating compensating differentials for on-the-job risk exposure in labor markets. Viscusi and Aldy (2003) provide an analysis of the extensive literature of VSL based on estimates using U.S. labor market data from the last three decades.

Half of the studies reviewed in Viscusi and Aldy (2003) provide estimates that range from \$5 million to \$12 million (in 2000 dollars). Estimates below \$5 million usually are reported by studies that use the Society of Actuaries data, which contains data on workers that self-selected themselves to jobs that are much riskier than the average job. On the other hand, studies that report estimates above \$12 million tend to estimate the wage-risk relationship indirectly. Viscusi and Aldy (2003) regard the median estimate of

\$7 million from the above table as the most reliable. These values of VSL using hedonic wage methods are similar to those generated by U.S. product market and housing market studies [44].

To transfer the estimates of VSL to non-labor market contexts, the preferences of the study population and the populations in the policy context must be similar. Other factors that may influence the transfer of VSL estimates are the age and income distribution of the study population. In addition to finding a positive association between income and VSL, Viscusi and Aldy (2003) also found a statistically significant relationship between union-status of workers and VSL. In the case of RPS, non-labor populations are impacted by the policy and may have a different VSL from that of the labor market.

Valuation of Morbidity Benefits

There are a number of health endpoints that can be quantified, but difficult to value in monetary terms [45]. Reduced lung function is an example. Currently, there are no studies available on the economic valuation of changes in lung function. One reason is that in many cases there is no clear connection between lung function and the economic well-being of an individual. Reduced lung function is typically associated with other symptoms, such as cough or asthma attacks, and it is unclear whether a temporary decrease in lung function would go unnoticed without the related symptoms, and hence it may have no economic value.

Several endpoints reported in the literature overlap with each other. For example, the various measures of restricted activity, or their definitions are not unique. Therefore, one must be careful not to include a combination of endpoints that could lead to double counting of benefits.

[43] Alternative measures to VSL include the Value of Statistical Life Years (VSLY) lost or saved. For example, if pollution abatement saves one person with average life expectancy of fifty more years, then we say that the policy results in fifty life years extended. VSLY can be interpreted as an age-specific VSL. Another alternative to VSL is the Quality Adjusted Life Years (QALY) measure. QALY adjusts life-years extended for the quality of life during those years. To estimate QALY, information is needed about the path and duration of health states and weights for the different health states must be chosen. See US EPA 1999.

[44] For example, Jenkins, Owens, and Wiggins (2001), Gayer, Hamilton, and Viscusi (2000), Dreyfus and Viscusi (1995), Carlin and Sandy (1991).

[45] Source: USEPA, "Review of the National Ambient Air Quality Standards for Ozone: Assessment of Scientific and Technical Information, Staff Paper, Office of Air Quality Planning and Standards, 1989. Quoting Avol et al. (1984), Kulle et al. (1985), McDonnell et al. (1983)

In a number of studies, it has been found that the different methods, mentioned in the previous sections, generate systematically different estimates for morbidity effects. For each effect, both estimates are from the same source. These results indicate that people's willingness to pay to avoid certain symptoms usually exceeds the cost of that symptom by an order of a magnitude. For instance, researchers have found that the willingness to pay to avoid nausea is \$72.30 whereas the individual cost of illness is \$3.78 (both in 1996 \$) [46]. The Appendix contains estimates of willingness to pay and cost-of-illness for various health effects.

Valuation of hospital admissions avoided

The typical approach for valuing the avoided incidence of hospital admissions is through the use of cost of illness (COI) method [47]. Well-developed and detailed estimates of hospitalization of the health effects are readily available [48]. In a complete cost-benefit analysis, COI estimates should be obtained for each health effects for which dose-response functions are available. Valuation estimates typically have two

components: cost of hospital stay, and lost earnings due to hospitalization. As mentioned above, COI method underestimates WTP, but there are currently no studies available that would estimate WTP directly.

Indirect Benefits to Humans through Ecosystems

Air pollution causes ecological effects, and Table 3.7 identifies the most important direct and indirect effects of air pollution.

The first step in valuing ecosystem benefits of reduced air pollution is to identify measurable effects. Freeman (1997) identified the following categories of ecosystem services to humankind:

1. Material inputs into economic activity (fossil fuels, minerals, animals)
2. Life-support services (breathable air, livable climate)
3. Environmental amenities used for recreation
4. Processing of waste products discharged into the environment

Table 3.7 Ecological Effects of Air Pollution

Pollutant Class	Major Pollutants and Precursors	Short-term effects	Long-term effects
Acidic Deposition	Sulfuric acid, nitric acid Precursors: SO ₂ , NO _x	Direct toxic effect to plant leaves and aquatic organisms	Progressive deterioration of soil quality and acidification of surface waters
Nitrogen Deposition			Saturation of terrestrial ecosystems with nitrogen. Progressive nitrogen enrichment of coastal estuaries.
Hazardous Air Pollutants (HAPs)	Mercury and dioxins	Direct toxic effects to animals.	Accumulation of mercury and dioxin in the food chain.
Ozone		Direct toxic effects to plant leaves.	Alteration of ecosystem-wide energy flow and nutrient cycling.

Source: USEPA (1999)

[46] Chestnut et al. (1998, 1996)

[47] See for example, US EPA (1999)

[48] For example, the Cost of Illness Handbook by EPA.

Economic analyses of air pollution control have paid less attention to ecological benefits than direct benefits to human health. There is a complex and nonlinear relationship between ecosystems and air pollution, and many impacts are difficult to measure. In addition, available valuation methods can measure only some of these benefits: material inputs and the value of environmental amenities used for active recreation.

The most important ecological benefits of air pollution abatement include:

- Eutrophication [49] of estuaries associated with atmospheric nitrogen deposition
- Reduced tree growth associated with ozone exposure
- Acidification of freshwater bodies associated with atmospheric nitrogen and sulfur deposition
- Accumulation of toxics in freshwater bodies associated with atmospheric toxics deposition
- Aesthetic damages to forests associated with ozone and airborne toxics

Table 3.8 Service Flows for Quantitative Assessment

Ecological Effect	Endpoint	Dose-Response Functions	Economic Model
Acidification	1. Forest Aesthetics 2. Recreational Fishing 3. Existence Value of Biodiversity	1. Not required 2. Multiple available 3. Multiple Available	1. Site-specific 2. Site-specific 3. Site-specific
Eutrophication	1. Recreational Fishing 2. Existence Value of Biodiversity		1. Site-specific 2. None available
Toxics Deposition	1. Forest Aesthetics 2. Recreational Fishing 3. Existence Value of Biodiversity 4. Hunting and Wildlife Aesthetics	1. Multiple available 2. Multiple available 3. Multiple available 4. Multiple Available	1. Site-specific 2. Site-specific 3. None available 4. Site-specific
Multiple Pollutant Stress	1. Ecosystem aesthetics and ecosystem existence value	1. None available	1. None available

[49] Eutrophication is a condition in an aquatic ecosystem where high nutrient concentrations stimulate blooms of algae. Increased eutrophication from nutrient enrichment due to human activities is one of the leading problems facing some estuaries in the Mid-Atlantic region.

Since not all ecological benefits are quantifiable or can be monetized, in valuation studies attention is often restricted to ecological impacts associated with benefits to humans, called service flows, rather than broad structural changes to ecosystems. The main criteria for including service flows in valuation studies are that they must be identifiable, quantifiable and monetizable. Table 3.8 summarizes service flows to humans that satisfy these criteria based on the literature cited in the Appendix.

There are four primary methods to monetize non-health-related benefits: hedonic property value models (similar in approach to hedonic methods applied to mortality benefits described previously), travel cost models (TCM), contingent valuation, and market models. TCM exploits observed differences between travel distance and environmental quality of recreation site to estimate the monetary value of each site characteristic. If people are willing to pay more in travel costs to enjoy an environmentally more pristine recreation site than a less pristine one, then the additional travel costs reflect the minimum value that person places on the more pristine site. Market Models study the impact of changes in ecological services on both producers and consumers of market goods that rely on these services, for example the impact on recreational fishing of decreasing toxic discharges in a lake. Contingent valuation methods ask people to express their monetary preferences for hypothetical changes in the environment, for example through the use of surveys asking consumers the additional costs they would be willing to pay to for improved visibility at a state park.

Agricultural Productivity Benefits

Air pollution has a negative impact on agricultural productivity. Research in the area has focused primarily on the economic impacts of tropospheric ozone, acidic deposition, and global climate change on commercial crops [50]. Economic assessments of the impact of air pollution on crop losses are sometimes associated with forestry impacts. However, as Spash (1997) points out, forestry is a multi-product system, in which the economic valuation of the impacts pertains to a much wider

set of issues, including biodiversity changes, reduced aesthetics and recreation services. Some of these impacts do not have commercial value, and therefore forestry damages are poorly represented in market-related production models that are commonly used to estimate agricultural productivity damages. Therefore, what follows is a review only of the methodology commonly used to estimate agricultural crop losses due to air pollution.

Pollutants that have been found to have a negative impact on crop yields are ozone (O_3) and its precursor pollutants (mostly NO_x), and sulfur dioxide (SO_2). Of all the pollutants, the most extensive research has been conducted on tropospheric ozone. Ashmore (1991) concludes that although gaseous pollutants other than ozone (namely SO_2 and NO_2) may be locally important at high concentrations, they have little economic impact on a national scale. Only minor damage to plants had been attributed to gaseous pollutants other than ozone and to sulfate and acid deposition. The National Crop Loss Assessment Network (NCLAN) program found a statistically significant response to SO_2 only in soybeans and tomatoes. Herrick and Kulp (1987) report a negligible impact of SO_2 and NO_2 within the National Acid Precipitation Assessment Program (NAPAP). Ashmore (1991) finds that barley, clover, and lucerne are especially sensitive to SO_2 , but these are minor crops associated with small potential economic benefits of pollution control.

Corn and soybean appear to be the most sensitive crops to acidic deposition. In addition, the available research seems to suggest that most commercial crop yields are relatively insensitive to acidic deposition on its own [51]. Spash (1997) concludes that crop damages from SO_2 , NO_2 and acid deposition combined are 5-10% of the crop damages of ozone pollution.

Segerson (1987) has identified a number of factors that distinguish the effects of acidic deposition from those of ozone [52].

[50] USEPA (1999)

[51] Segerson (1991) pp. 352

[52] This is the most recent research that is available.

- Acidic deposition affects a wide range of non-market goods while ozone affects mostly commercial crops
- Acidic deposition is a dynamic pollutant that accumulates over time, while ozone is periodic. Therefore an economic analysis of ozone exposure may be based on short-term studies, while acidic deposition should be based upon assessing accumulated impacts over time.
- The impacts of acidic deposition are surrounded by a greater degree of uncertainty than those of ozone.
- Ozone pollution is a more localized problem than acidic deposition.

Ozone has been observed to cause significant damages in terms of crop yield losses at current ambient levels. Furthermore, the increased frequency and duration of hot dry weather implied by global warming will increase the concentration of tropospheric ozone from available precursors. Table 3.9 below illustrates the damages to crops from ozone exposure.

The majority of economic assessments of ozone damage to crops have been at the regional level. Published studies have concentrated on two main regions of the U.S., namely, the Corn Belt (Illinois, Indiana, Iowa, Ohio and Missouri) and California.

To quantify agricultural productivity benefits, dose-response functions that describe the relationship between ambient ozone concentrations and changes in crop yields are used. Estimated minimum and maximum dose-response function for six major crops are described in Appendix, Section 5.1.

Valuation of Agricultural Productivity Benefits

The traditional approach of valuing crop losses due to air pollution was to calculate monetary equivalents of the approximated losses by multiplying decreased yields by the current market price to give a producer loss estimate equal to total revenue. This method is likely to overestimate the gain to producers from ozone reductions because it ignores farmers' reactions in terms of changing the input mix and cross-crop substitution [53].

Table 3.9 Processes and Characteristics of Crop Plants That May Be Affected By Ozone

Growth	Development	Yield	Quality
Rate	Fruit set & development	Number	Appearance: size, shape, and color
Pattern	Branching	Mass	Storage life
	Flowering		Texture/cooking quality, Nutrient content, Viability of seeds

Source: Jacobson (1982) p. 296, Table 14.1.

Although ozone-induced quality degradation may be a significant part of total economic damages, research has almost exclusively focused on estimating changes in output resulting from air pollution. There is currently insufficient information available as to the importance of crop quality response (Spash, 1997).

In more recent empirical work agricultural production models have been used to estimate the economic costs associated with yield losses due to air pollution. These models estimate the social benefit from reduced ozone damages.

Various agricultural production models have been used in economic assessments of ozone damage.

[53] Spash (1997)

Recreational and commercial fishing benefits form a subgroup of ecological benefits of air pollution. The theoretical basis for valuing ecological benefits in general and recreational and commercial fishing benefits in particular, is that the natural environment provides us with services that we value [54]. There are no suitable methods to comprehensively value many of these service flows (e.g. breathable air, livable climate). Therefore in valuation studies, we are limited to valuing services flows that are either material inputs into our economy, or provide amenities associated with marketed services (e.g. recreation).

Three types of pollution are associated with commercial and recreational fishing: acidification, nitrogen eutrophication, and toxics deposition. Acidification, or acid deposition, is probably the best-studied effect of air pollution on ecosystems. The main cause of acidification is acidic precipitation in the form of sulfuric acid (H_2SO_4) and nitric acid (HNO_3). These acids are formed from sulfur dioxide (SO_2) and nitrogen oxides (NO_x) found in the atmosphere. Electric power plants are among the primary point sources of SO_2 . On the other hand, a large share of NO_x is from non-point sources (transportation), and therefore anthropogenic NO_x is more dispersed in the atmosphere compared with anthropogenic SO_2 .

Deposition occurs via three main pathways: (1) wet deposition, where the pollutant is dissolved in precipitation, (2) dry deposition, which is a direct form of deposition of gases and particles to any surface, and (3) cloud-water deposition, when cloud or water droplets are intercepted by vegetation. Since most of the precipitation falls on the terrestrial part of the catchments, soil properties are generally strongly associated with water quality. Consequently, acidification resulting from acid deposition usually occurs in areas with acidic soils. Throughout the world freshwater acidification is the most serious in eastern parts of the United States. Reductions of anthropogenic SO_2 and NO_x emissions in Europe in recent years have resulted in an improvement in acidified water bodies, however, the same trend has not been observed in the United States [55]. It is

believed that it may take ecosystems several decades to recover from the impact of acidification even after emissions have been cut.

Eutrophication is the result nitrogen deposition leading to excessive nitrogen enrichment of aquatic ecosystems, and it may adversely affect the biogeochemical cycles of watersheds. Atmospheric nitrogen is deposited into water bodies through dry and wet deposition. Excessive eutrophication can lead, for example, to massive algae blooms, which in turn reduces the oxygen levels and leads to habitat loss. It is estimated that 86% of the East Coast Estuaries are susceptible to eutrophication [56].

Toxics deposition involves hazardous air pollutants (HAPs), as defined by the Clean Air Act: mercury, polychlorinated biphenyls (PCBs), chlordane, dioxins, and dichlorodiphenyltrichloroethane (DDT). When considering air pollution from power plants, the most important of these pollutants is mercury. Much of mercury found in ecosystems comes from natural sources. Mercury accumulates in fish, birds, and mammals, and it may be dangerous to humans when the concentration exceeds a critical level. Mercury-based statewide fish consumption advisories are fairly common in the United States.

Table 3.10 below summarizes the main recreational and commercial fishing impacts associated with acidification, eutrophication, and toxics deposition.

Following emissions modeling, and transport and deposition modeling, the next step in the process of quantifying the benefits to recreational fisheries is the use of an exposure model. Unfortunately, comprehensive models to quantify the impacts of acidification, eutrophication, and toxics deposition are currently not available.

To quantify the impact of acid deposition on fisheries, US EPA (1999) uses a region-specific model to quantify the effects of acidification on freshwater fish populations: Model of Acidification

[54] Freeman (1997)

[55] Stoddard et al. (1999)

[56] US EPA (1997)

Table 3.10 Recreational and Commercial Fishing Can Be Associated With the Following Ecological Impacts of Air Pollution

Pollutant Class	Ecosystem Effect	Service Flow
Acidification (H ₂ SO ₄ , HNO ₃)	Freshwater acidification resulting in fish (and other aquatic organism) decline	Recreational Fishing
Nitrogen Eutrophication (NO _x)	Freshwater acidification resulting in fish (and other aquatic organism) decline	Recreational Fishing
Toxics Deposition (Mercury, Dioxin)	Aquatic bioaccumulation of mercury and dioxin	Recreational and Commercial Fishing

Source: US EPA (1999)

of Groundwater Catchments (MAGIC) [57]. MAGIC is calibrated to the watershed of an individual lake or stream and then used to simulate the response of that system to changes in atmospheric deposition. The model simulates the effects of acid deposition on both soils and surface waters. The simulation typically involves seasonal or annual time steps and is implemented on decadal or centennial time scales.

Quantifying Eutrophication

When atmospheric nitrogen is deposited in estuaries, it can lead to eutrophication. Estimation of a dose-response relationship between nitrogen loading and water quality changes is complicated because of the dynamic nature of ecosystems. Most likely, these dose-response functions are nonlinear with a threshold. Unfortunately, universally transferable dose-response functions for quantifying the effects of eutrophication have not yet been developed. USEPA (1999) study quantified deposition-related nitrogen loadings for three estuaries (Chesapeake Bay, Long Island Sound, Tampa Bay) using GIS-related methods. Data on nitrogen deposition, together with information on abatement options to reduce excess nutrient loads, was used for valuation purposes. In addition, USEPA (1999) used specific biophysical indicators of estuarine health to quantify the benefits. This approach is useful when there is a direct link between the biophysical indicator and the ecological service flows. USEPA (1999) used the properties of the seagrass bed, which provides habitat for variety organisms, and have been shown to decline with increased nitrogen deposition, as an indicator.

[57] Cosby et al., (1985a,b)

Quantifying Toxics Deposition

Most damages to ecosystems are caused by five hazardous air pollutants (HAP): mercury, PCBs, dioxins, DDT, and chlordane. The mechanism of ecosystem responses to toxic contamination is poorly understood. Furthermore, service flow impacts of ecosystem damages are difficult to observe because HAPs persist in aquatic ecosystems for a long time. A comprehensive quantitative analysis with the available models and data is currently not possible.

Valuation of Recreational and Commercial Fishing Benefits

Unlike commercial fishing, recreational fishing is to a large part a non-market activity. Although most states charge a license fee for recreational fishing, the license fee itself is believed not to reflect the true willingness-to-pay (WTP) for recreational fishing. Marginal WTP for recreational fishing is a function of catch attributes (e.g. number and the average size of fish caught) and other determinants. Environmental factors indirectly affect WTP for recreational fishing by affecting the catch attributes. The total value of recreational fishing to the angler can be measured by the consumer surplus, which is the difference between WTP and the actual amount they pay or the cost they incur for the recreational fishing day. Consumer surplus is measured by the area below the demand curve and above the price or the cost of a recreational fishing day.

There is an extensive literature on valuation of fishing opportunities by anglers. In the valuation

literature, two primary methods have been used most often to deduce the value of recreational fishing: travel cost method (TCM), and contingent valuation method (CVM). TCM uses observed travel costs to visit a fishing site and per-capita visitation rates to deduce the demand for recreational fishing. On the other hand, CVM is questionnaire-based method, where anglers are asked hypothetical question about how much they would be willing to pay for a day of fishing.

TCM cannot be used to measure willingness-to-accept (WTA) some degradation in an environmental amenity (i.e. compensation demanded for an environmental damage). Hence, TCM cannot be used to estimate the value of loss of fishing opportunity due to air pollution. Furthermore, the use of CVM in general has generated controversy in the valuation literature. CVM is subject to an inherent bias due to its hypothetical nature. Study participants are often subjected to an unfamiliar market context, or they may not be fully aware of the characteristics of the good in question, or their own budget constraints. Some critics of CVM have pointed out that estimates of WTP obtained using CVM may not reflect the true WTP for the non-market good, but they rather reflect the WTP for moral satisfaction. The answers from CVM studies may be biased because of passive-use motives, such as the “warm glow” effect [58]. Individuals’ responses to WTP questions serve the same function as charitable contributions, and people are assumed to get a “warm glow” from giving. Some economists do not fully recognize “warm glow” as an economic value. Snyder et al. (2003) obtain estimates of the value of a recreational fishing day for 48 U.S. states. In the Appendix, summarizes the results for Mid-Atlantic states that are most likely to be affected by the New Jersey RPS are summarized.

A comparison of Snyder et al. (2003) estimates with the values from other studies reveals that there is a considerable geographic variation in the estimated value of recreational fishing. Moreover, the estimates are significantly lower than those reported in other studies employing TCM or CVM. The differences could be due to methodological differences, as well as, to the elimination of the biases in TCM and CVM. Ranges vary from a few dollars a day to almost twenty.

[58] Andreoni (1989)

Another limitation of many early studies is that they do not include a direct measure of water quality. A notable exception is Montgomery and Needelman (1997) that consider the special case of toxic contamination of fisheries. Toxic contamination is a special case of pollution, because contaminants in fish become dangerous to humans eating fish before they result in a decline in fish population. In addition, through health advisories the public is better informed about incidences of toxic contamination than other forms of pollution (e.g. acidification or eutrophication).

Biodiversity Benefits

Biodiversity is a valuable environmental benefit. A number of human actions, including anthropogenic air pollution, have led to a dramatic decline in biodiversity across the globe (Pimm et al., 2001). Biodiversity refers not just to an accumulation of species in a given area, but it also incorporates the ecological and evolutionary interactions among them (Armsworth et al., 2004).

The first step in estimating biodiversity benefits is defining biodiversity. Biodiversity encompasses four levels, which are summarized in Table 3.11.

Genetic biodiversity is the most basic level, and it refers to the information represented in the DNA of living organisms. Species-level biodiversity refers to the variety of species in a given area. Because only a small fraction of the estimated 5-30 million species currently living on the earth (Wilson, 1988) have been identified and described, empirical estimates of the species-level biodiversity are often surrounded by a great degree of uncertainty. Community-level biodiversity is important, because it is believed that species-level diversity enhances the productivity and stability of ecosystems (Nunes and van den Bergh., 2001, Odum, 1950). However, recent studies suggest that no pattern or determinate relationship may exist between species-level diversity and stability of ecosystems (Nunes and van den Bergh. 2001, Johnson et al. 1996). Functional diversity, or the ecosystem’s functional robustness, refers to the ability of the ecosystem to absorb external shocks. Unfortunately, the ecosystem’s functional diversity is still poorly understood (Nunes and van den Bergh, 2001).

Table 3.11 Four Levels of Biodiversity

Type of Biodiversity	Physical Expression
Genetic	Genes, nucleotides, chromosomes, individuals
Species	Kingdom, phyla, families, genera, subspecies, populations
Ecosystem	Bioregions, landscapes, habitats
Functional	Ecosystem, functional, robustness ecosystem resilience services goods

Source: Turner et al. (1999)

Human threats to biodiversity include activities causing habitat loss (conversion, degradation or fragmentation) and climate change, harvesting, as well as the introduction of exotic species that by becoming dominant competitors or effective predators may drive many native species to extinction.

Empirical estimates of Morse et al. (1995) and Field et al. (1999) of the impact of climate change on biodiversity illustrate the magnitude of threats to biodiversity: 4 °F average temperature increase can reduce the number of all species in California by 5%-10%.

A traditional approach to measuring biodiversity has focused on species-level biodiversity, which can be measured in two ways [59]: richness – number of species in a given area – and evenness – how well distributed abundance or biomass is among species within a community. Evenness is the greatest when species are equally abundant. For example an area that has a total population of 100 of 10 different species, each comprising of 10 individuals, is more diverse than a community of 1 species with 91 individuals and 9 other species with one individual each. To quantify diversity, a diversity index may be used that combines aspects of richness and evenness.

The monetization of biodiversity benefits requires assessing what it is about biodiversity that consumers' value. In general, consumers' benefit can be divided into *use value* (direct such as tourism or indirect such as pollination) or *non-use value* (intrinsic

or existence value). Direct benefits to consumers arise in two important ways:

- **Service flows** – Ecosystems provide valuable services to society, such as water purification in natural watersheds, prevention of soil erosion and carbon sequestration by standing forests, and recreational services such as ecotourism and birdwatching. The service flow approach to valuation is predicated upon investments in preserving or restoring biodiversity needed to deliver a competitive return relative to other investment opportunities within the economy otherwise capital would be put to other uses, and hence it does not necessarily maximize some types of diversity (Armsworth, 2004).
- **Bet hedging** – Conserving biodiversity provides society with a hedge against unforeseen circumstances. For example, if society were overly reliant on monocultured ecosystems, it would be vulnerable to catastrophic losses due to disease outbreaks.
- **Nonuse or existence value** – Consumers derive utility from knowing that certain species (still) exist.

Nunes and van den Bergh (2001) critically evaluate a number of biodiversity valuation studies at each level of biodiversity value. The authors conclude that available economic valuation estimates should be regarded as providing an incomplete perspective on the value of biodiversity changes, and they provide at best the lower bounds on that value. The main shortcoming of these single-species valuation studies is that they do not account for species substitution and complementary effects. Multiple-species valuation

[59] Much of the following discussion is based on Armsworth et al. (2004).

studies account for all related species, and the resulting estimates are in general higher than those of single-species studies. There are also studies that link the value of biodiversity to the value of natural areas with high tourism and outdoor recreation demand. Studies reviewed by Nunes and van den Bergh (2001) report a wide range of estimates for the various levels of biodiversity valued from hundreds of thousands of dollars to hundreds of millions depending on the biodiversity level.

Appendix B presents tables that summarize the results from the valuation studies that have been performed in North America.

Forestry Benefits

Air pollution has been recognized as a potential problem for forests for a long time. Sulfur dioxide, fluorides, heavy metals and ozone pose the greatest threat to forest ecosystems. In the past, sulfur emissions, that cause acid rain, were the primary concern, but in recent decades massive efforts to reduce this pollutant have been largely successful. Today, in terms forestry impacts, ozone may be the pollutant associated with the greatest potential benefits.

Scientific evidence suggests that elevated tropospheric ozone levels disrupt vegetation growth, and interfere with the respiratory function of plants carried out by photosynthesis even at concentrations below current air quality standards [60]. Sometimes ozone injury to plants has observable effects such as yellowing or stippling of leaves, but negative impacts of ozone often occur without accompanying visible symptoms.

Ambient ozone enters the plant through pores in the leaf or needle called stomata, where most of the plant's metabolic and respiratory activity occurs. Once ozone enters these stomata, it initiates a chain reaction that destroys or damages plant proteins and enzymes, as well as the fatty chemicals that help form cell membranes. Plants continue to suffer damage long after the ozone exposure episode is over. Furthermore,

some researchers have suggested that there are synergies between ozone and acid deposition [61] .

Ozone damage to forests is a common problem in many parts of the eastern U.S. Particularly sensitive species to ozone are the poplars (*Populus* spp.), white pine (*Pinus strobus*), and the oak family (*Quercus* spp.).

Another serious threat to forest ecosystems is acid deposition in the form of nitrogen acids due to nitrogen oxides emissions. Aluminum is naturally present in forest soils in the form of chemical compounds that are harmless to living organisms. Nitrogen acids cause ions of aluminum to become mobile in soil, and in its toxic form, aluminum is taken up into the tree's roots. This may result in reduced root growth, which reduces the tree's ability to take up water and withstand drought. Excess nitrogen is also absorbed directly from the air through the leaves during fog and low clouds. If ozone is present in sufficient concentrations, exposure to this oxidant can damage the leaves, damaging respiration processes of the organism.

Given the evidence on damage to forests and the size of the forest cover, forestry benefits seem to play an important role in total benefits due to reduced air pollution in the northeastern United States. According to Mid-Atlantic Integrated Assessment (MAIA) – the multi-agency effort headed by the USEPA to assess the health and sustainability of ecosystems – forests cover 61% of the total land area in the MAIA region [62]. Ninety-five percent of the region's forests are classified as timberland. The vast majority (79%) of timberland is owned by nonindustrial private landowners, while the forest industry owns approximately 7%. Hardwood forests dominate the MAIA region. For the region as a whole, oak/hickory is the predominant forest type. Other dominant forest types in the region are northern hardwoods, loblolly/shortleaf pine, and oak/pine.

Due to the different life cycles involved, the assessment of forest damage is substantially more

[60] See for example, Wang et al. (1986) and Reich and Amudson (1985)

[61] Hewitt (1990)

[62] The MAIA study region includes Delaware, Maryland, Pennsylvania, Virginia, and West Virginia, and parts of New Jersey, New York, and North Carolina.

difficult than that of agricultural crop damage. On one hand trees live for a long time, which makes the study of pollution impacts much more difficult. On the other hand, unlike agricultural soils, which are effectively managed through annual cycles, forest soils are much less disturbed which leads to an accumulation of acidification impacts.

USEPA (1999) uses the PnET-II model to estimate the impacts of tropospheric ozone on commercial timber growth. The PnET model simulates the cycles of carbon, water, and nitrogen through forest ecosystems. Model inputs of monthly weather data and nitrogen inputs are used to predict photosynthesis, evapotranspiration, and nitrogen cycling on a monthly time-step for several forest types.

Valuation of Forestry Benefits

The valuation techniques of forestry benefits can be grouped into three categories: 1. direct market prices, 2. indirect market prices, 3. hypothetical values. Methods using direct market prices are based on actual prices, and consequently they do not reflect some benefits (e.g. preservation of biodiversity) that the market participants did not take into account in their decision-making. Methods utilizing indirect market prices include hedonic property values, the travel costs, opportunity costs, surrogate prices and replacement costs [63]. The opportunity cost method uses the market price of the best alternative forgone to provide a lower bound on forestry benefits. Surrogate prices methods use the market price of a close substitute as a proxy for the benefits. A surrogate market approach is used by methods using hypothetical values. Two methods that belong to this category are the contingent valuation method, and conjoint analysis. As described in the previous sections, contingent valuation method uses surveys that ask hypothetical questions to estimate economic values. Conjoint analysis estimates values by asking people by asking people questions across a range of features or attributes of a forest [64].

Forestry benefits may be grouped into three categories for valuation purposes: on-site private

benefits, on-site public benefits, and global benefits. On-site private benefits include timber productions, agricultural and other agroforestry products, and non-timber forest products (e.g. mushrooms, medicinal plants, honey, fruits, nuts, etc.), and recreation and tourism. On-site public benefits include watershed protection, agricultural productivity enhancement, nutrient cycling, microclimate regulation, and aesthetic, cultural, and spiritual values. Global benefits include carbon sequestration, and biodiversity conservation. Values derived for some of these benefits are not transferable, and therefore most valuation studies restrict attention to on-site benefits such as timber production. These benefits are usually estimated using market models. For example, USEPA (1999) used the USDA Forest Service Timber Assessment Market Model to estimate market changes that result from reduced timber growth.

Indirect Benefits to Humans through Nonliving Systems

There are two types of indirect benefits through nonliving systems that were studied the most: avoided materials damages, and improved visibility. The steps of estimating these benefits are summarized in the sections that follow.

Avoided Materials Damages

Anthropogenic sulfur and nitrogen pollutants are believed to have caused vast damage to buildings, structures, as well as the cultural heritage in the past century. Much of the damage has occurred in Europe and North America, but with growing car traffic, and very high concentrations of sulfur dioxide in many cities of China, India, and Latin America material damages due to air pollution continue to remain a significant problem.

Objects most susceptible to air pollution are the ones with long lives, particularly buildings. Other objects, such as cars, may be damaged by air pollution, but these damages tend to be less important, because they are usually replaced before the damage could become significant.

[63] Cavatassi (2004)

[64] Cavatassi (2004)

Pollutants that contribute to degradation of buildings are particles (particularly soot) causing soiling, and sulfur dioxide (SO₂) contributing to corrosion and erosion caused by acid rain. Principal effects associated with air pollution are loss of mechanical strength, leakage, failure of protective coatings, loss of details in carvings, and pipe corrosion.

SO₂ has a strong accelerating effect on the degradation of certain materials by contributing to corrosion by acidic deposition. Atmospheric corrosion is influenced by climatic patterns such as relative humidity, temperature and precipitation, and it tends to be a local problem, because the damage often occurs near the source of emission. On the other hand, indirect effects of SO₂ emissions caused the acidification of soil and water bodies tend to be a regional problem due to the long-range transport of air pollutants.

Air pollution damages materials such as zinc, copper, stone, as well as organic materials. In case of zinc and copper, the dissolution of protective corrosion products leads to increased deterioration rates. Calcareous stones, such as limestone or marble, are very susceptible to acid deposition by sulfur dioxide through transformation of the original calcium carbonate to gypsum and calcium sulfate. Degradation of organic materials, such as rubber tires and paints, are usually associated with ozone in conjunction with temperature and solar radiation.

The ideal approach to quantifying materials damages is analogous to the approach used to quantify health endpoints. One would start by estimating the change in pollutant concentrations caused by a policy-induced reduction in emissions. The second step would involve the use of dose-response or concentration-response (CR) functions that relate the physical damage to ambient pollutant concentrations. And the last step involves attaching monetary values to damages.

A valid CR function provides a mathematical relationship between properties of the environment and some index of materials, such as loss of stock thickness. Some early attempts aimed at deriving such CR functions focused on the relationship between

ambient pollutant concentrations and corrosion rates. As Lipfert (1996) points out, this approach neglects the important variable of delivery of the reactant to the surface. A full understanding of the process requires a separation of pollutant delivery process from the subsequent chemical reactions. The appropriate technique to estimate a CR function is that of a multiple regression, in which some index of corrosion is the dependent variable and the various environmental factors are the independent variables.

Perhaps the most difficult element of economic assessment of materials damages (or benefits from reduced air pollution) is the problem of estimating stocks of buildings at risk [65]. One problem is that there is considerable heterogeneity in the use of housing materials across country. While residential housing materials tend to follow regional patterns, commercial and industrial buildings tend to be more uniform. Another important problem since atmospheric corrosion has been present in many parts of the country for a long time people may have substituted away from the more sensitive building materials toward less sensitive ones. The greatest difficulty lies in distinguishing chemical characteristics of exposed surfaces within each building type and category. Lipfert (1996) suggests that there is a need for a probabilistic, as opposed to deterministic, approach to assessment. There are many relevant but disparate databases on building stocks, but no effort has been made yet to synthesize that information [66].

As an alternative to the above bottom-up approach, Rabl (1999) estimates damages to buildings in France by working with aggregate data on observed frequencies of cleaning and repair activities. The result is a “combined concentration-response function”. The main variables in the CR function are income and the ambient concentration of particulate matter.

Rabl (1999) considers two types of damage caused by air pollution: corrosion or erosion of coatings and construction materials and soiling. Corrosion and erosion are primarily due to acid deposition. A large number of studies analyzed the effects of air pollution, corrosion, and erosion. Dose-response functions have

[65] Lipfert and Daum (1992)

[66] Lipfert (1996)

been estimated for several building materials [67]. There are relatively few studies on soiling due to air pollution, and consequently few dose-response functions are available [68].

Using a bottom-up approach, the following are the steps in valuations:

- Division into pollution strata
- Materials inventory and inspection of physical damage
- Damage functions
- Estimated change in service life
- Maintenance/ Replacement cost
- Estimated economic damage

The main drawback of the bottom-approach is the need for very detailed data on building inventories. Neither approach to valuation may be used to assess damages to the cultural heritage. Cultural heritage encompasses both outdoor buildings and sculptures and treasured objects kept indoors, stored in museums and archives. The most appropriate valuation method for assessment is contingent valuation. These valuation studies tend to be case specific and generally not transferable.

Visibility Benefits

Reduced visibility due to anthropogenic air pollution affects some of the country's most scenic areas. US EPA estimates that in national parks in the eastern United States, average visual range has decreased from 90 miles to 15-25 miles. In the West, visual range has decreased from 140 miles to 35-90 miles. The main cause of visibility impairment is haze. Under stagnant air mass conditions, aerosols can be trapped and produce a visibility condition usually referred to as layered haze. Some light is absorbed by particles while other light may be scattered away before it reaches the observer. The introduction of particulate matter and certain gases into the atmosphere therefore reduces visibility.

From a technical point of view, visibility is a complex and difficult concept to define. Visibility

includes psychophysical processes and concurrent value judgments of visual impacts, as well as the physical interaction of light with particles in the atmosphere. Therefore it is important to understand the psychological process involved in viewing a scenic resource, and to be able to establish a link between the physical and psychological processes.

Quantifying visibility requires developing links between visibility and particles that scatter and absorb light. Visibility, in the most general sense, reduces to understanding the effect that various types of aerosol and lighting conditions have on the appearance of landscape features. Measuring visibility by a single index, however, is not possible because visibility cannot be defined by a single parameter [69]. Many visibility indices have been proposed, however, the most simple and direct way of communicating reduced visibility is through a photograph. In fact, many contingent valuation (CV) studies of visibility present the subjects with photographs of scenic areas with varying levels of visibility. The reason photographs communicate visibility-changes so well is that the human eye works much like a camera. The human eye detects relative differences in brightness rather than the overall brightness level, that is to say, the eye measures contrast between adjacent objects.

Because the human eye functions like a camera, a photograph captures visibility changes, as humans perceive it. However, it is difficult to extract quantitative information from photographs, and therefore direct measures of fundamental optical measures of the atmosphere have been developed. The most common measures are atmospheric extinction and scattering.

The scattering coefficient is a measure of the ability of particles to scatter photons out of a beam of light, while the absorption coefficient is a measure of how many photons are absorbed. Both coefficients are expressed as a number proportional to the amount of photons scattered or absorbed per distance. The sum of scattering and absorption is referred to as extinction or attenuation.

The most commonly used methods for visibility valuation are hedonic property values and contingent

[67] See, for example, Kucera (1990), Haneef et al. (1992), Butlin et al. (1992), and Lipfert (1987)

[68] See Hamilton and Masnfield (1992)

[69] Malm (1999)

valuation. Hedonic methods are based on revealed preference of consumers, because they link nonmarket valuation to a traded commodity. Hedonic property value studies estimate the marginal WTP function on the basis of an estimated relationship between housing prices and housing attributes (including air quality).

There are several factors that affect the relationship between property values and air quality. These include adverse health effects, reduced visibility or soiling due to air pollution. Hedonic methods cannot be used to estimate individual effects separately. Disaggregation of overall impacts requires making subjective judgments by the researcher. Nevertheless, the results of hedonic property value studies confirm the hypothesis that air quality has a significant impact on property prices. Kenneth and Greenstone (1998) estimate that the Clean Air Act induced nationwide monetized benefits were \$80 billion (in 1982-84 dollars) in the 1970's, and \$50 billion during the 1980s. Delucchi, Murphy and McCubbin (2002) estimate monetized costs of total suspended particle pollution in 1990 at \$52-\$88 billion in (1990 dollars). Some studies [70] attempted to disaggregate property value impacts into health, visibility, soiling, and other impacts. They find that visibility impacts are the second most important, after health effects, representing 19-34% of total monetized benefits.

Burtraw et al. (1997) present the results of an integrated assessment of the benefits and costs of the Title IV of the 1990 Clean Air Act Amendments initiated reductions in emissions of sulfur dioxide and nitrogen oxides. They use the Tracking and Analysis Framework (TAF) developed for the National Acid Precitation Assessment Program (NAPAP). Although uncertainties surround their estimates, the findings suggest that the benefits of the program substantially outweigh its costs. Two types of visibility effects are examined: recreational visibility at two national parks

(Grand Canyon and Shenandoah), and residential visibility in five metropolitan areas (Albany, NY, Atlantic City, NJ, Charlottesville, VA, Knoxville, TN, and Washington, DC). The results, summarized in Table 3.12, are most usefully considered on a per capita basis.

Table 3.12 Per Capita Benefits in 2010 for Affected Population

Effect	Benefits per Capita (1990\$)
Morbidity	3.50
Mortality	59.29
Aquatic	0.62
Recreational Visibility	3.34
Residential Visibility	5.81
Costs	5.30

Source: Burtraw et al. (1997), Table 2, pp. 13

These visibility estimates illustrate their potential magnitude, but it should be noted that they are based on the relatively small number of studies available in the literature, and also the geographical scope of the project is limited. Burtraw et al. (1997) explain the relatively large magnitude of visibility benefits compared to other types of benefits, namely aquatics, by claiming that willingness to pay depends on the availability of substitutes, and visibility, along with health, has no close substitutes.

Smith and Osborne (1996) perform a meta-analysis of visibility valuation studies to test whether CV estimates of WTP are responsive to the amount, or scope, of the environmental amenity being offered. They consider an internal consistency test for CV-based WTP. Internal consistency tests assess the reliability and validity of CV surveys. One way to evaluate the CV method is to compare WTP functions estimated with CV surveys with the specific, observable properties that economic theory implies WTP should follow. Smith and Osborne (1996) selected five of CV studies that used comparable methods for the meta-analysis. These studies focused on air quality as a key element. Furthermore, in each study air quality is presented in a way that permits computation of the change in visible range. The five selected studies are summarized in Table 3.13.

[70] For example, see Brookshire et al. (1979; 1982), Loehman et al. (1994), and McClelland et al. (1991)

Table 3.13 Summary of CV studies for visibility at national parks analyzed by Smith and Osborne (1996)

Authors	Mean and inter-quartile range of WTP (per month in 1990 \$)	Mean change in visibility	Location	Type of survey
Rowe et al. (1980)	\$9.27 (\$6.83, \$10.82)	0.50	Navaho Recreation Area	In-person interviews administered to to households in area
MacFarland et al	\$2.75 (\$1.69, \$3.73)	1.18	Grand Canyon and Mesa Verde National Parks	In-person interviews administered to visitors to the area
Schulze et al.	\$8.50 (\$4.42, \$11.67)	0.79	Grand Canyon, Mesa Verde, and Zion National Parks	In-person interviews administered to households in Albuquerque, Los Angeles, Denver, and Chicago
Chestnut and Rowe	\$4.35 (\$3.15, \$5.48)	0.62	Grand Canyon, Yosemite, and Shenandoah National Parks	Mail with telephone households in Arizona, Virginia, California, New York, and Missouri
Balson et al.	\$0.46 (\$0.007, \$0.97)	0.955	Grand Canyon National Park	In-person interviews conducted in St. Louis and San Diego Counties

Source: Smith and Osborne (1996), pp. 291, Table 1

The findings of Smith and Osborne (1996) support a positive, statistically significant and robust relationship between the WTP estimates and the percentage improvement in visible range. These results suggest that it may be possible to transfer results from a meta-analysis of past CV studies. The crucial issue in benefit transfer is to find a common metric to measure the environmental amenity.

IV. Conclusion

The proposed New Jersey RPS will reduce emissions generated by power plants that adversely affect the health and welfare of its population. Translating reduced emissions into reduction of emission concentrations, then into reduced health effects, and finally into a monetary value requires extensive New Jersey specific data, modeling, and analysis. Policy makers, however, must make decisions before such a research effort can be completed and should not ignore environmental externalities because

of the difficulty in quantifying them. One option is to apply environmental externality adders, which other policymakers have done, to the New Jersey proposed RPS. The illustrative calculations in this Chapter indicate that the environmental benefits of a

20% RPS are in the range of several hundred million dollars. Chapter 4 then discusses the associated policy implications of the proposed RPS.

Chapter 4: Summary of Findings and Discussion of Policy Recommendations

Assuming cost reductions in renewables occur as expected, the impact of the establishment of a 20% RPS in 2020 on the New Jersey economy is negligible. The impact may be positive under scenarios of higher fuel prices. Moreover, there are other benefits of the proposed 20% RPS that can be realized if the state takes action to implement a variety of policy initiatives. After summarizing the findings of the economic assessment of the proposed New Jersey 20% RPS, this chapter discusses initiatives and policy recommendations that should be considered by state officials in order to capture fully the benefits of increasing the RPS.

I. Summary of Findings

The major findings regarding the economic impact of the proposed 20% RPS compared to the existing RPS are the following:

1. Under the Expected Case assumptions, the proposed 20% RPS compared to the existing RPS would raise electricity prices approximately 3.7% in the year 2020 and have negligible impact on the growth of New Jersey's economy;
2. If natural gas prices rise to levels assumed in the High Energy Price scenario, the proposed 20% RPS has a positive economic impact on the New Jersey economy because electricity prices would be lower than under the existing RPS scenario;
3. Under the proposed 20% RPS, the location in New Jersey of all of the manufacturing, operations, and maintenance facilities and employees needed to support the PV and off shore wind infrastructure in New Jersey would be adding approximately 11,700 jobs and attenuate economic benefits to the New Jersey economy in the year 2020. In addition, the number of jobs could grow as demand for goods and services extends throughout the region;
4. The health and environmental benefits from a RPS result from and depend on the reduction in the atmospheric concentration of emissions, which are (with the exception of carbon dioxide) geographic specific, and other policymakers use a wide range of externality values that may have limited application to New Jersey but these values are used in this report for illustrative purposes;
5. Illustrative calculations using generic environmental externality adders indicate that in the year 2020 several hundred million dollars in environmental damage may be avoided by implementing a 20% RPS;
6. The proposed 20% RPS would lower natural gas prices for consumers in New Jersey and nationwide by reducing the burning of this fuel in power generation;
7. The proposed 20% RPS would increase reliability by providing electricity when grid power is not available and may reduce expenditures on T&D within the state;
8. The economic and electricity price impacts of the proposed 20% RPS depend substantially on whether expected cost reductions occur that reduce the cost of PVs and wind power; and

9. Existing cap-and-trade emission allowance policies for sulfur dioxide and nitrogen oxide act in combination with a RPS so that the RPS may not alone result in reduced levels of these emissions but will lower the price of emission allowances.

The proposed 20% RPS has many important policy implications, and New Jersey's ability to obtain the full benefits of the proposed 20% RPS depend on future actions. Recommendations are divided into two major categories – those that New Jersey can act upon unilaterally and those that require New Jersey to coordinate with other states. These recommendations are also discussed in the context of making connections between them and other policies and discussing those that improve knowledge by which to better inform policy.

II. Recommendations that New Jersey Can Implement Unilaterally

- **Monitor future cost reductions in renewable technologies particularly wind and PVs.** The economic and electricity price impacts of the proposed 20% RPS depend substantially on whether expected technological improvements and other factors occur that reduce the cost of PVs and wind power. The costs of these technologies are the key driver of the effects on electricity prices and economic growth under the proposed 20% RPS. The BPU should track and reevaluate the costs of wind, PVs, and other renewables to ensure that policies are being pursued to lower the cost of these renewable technologies and adjusted to their changing cost structure. If costs reductions occur slower than anticipated, consideration should be given to adjusting downward the level of renewables required. Similarly, if cost reductions occur more rapidly, thought should be given to accelerating the proposed 20% RPS. This cost monitoring should occur in tandem with setting the alternative compliance payment (ACP), the payment that suppliers must pay if they do not have sufficient renewables in their portfolio. The ACP performs both the role as an incentive to have suppliers procure required quantities of renewables but also acts to mitigate market power

in the REC market by, in effect, capping the price of RECs. Accordingly, the BPU should require the committee that advises it as to the setting of ACP to include a specific report on actual and projected cost of renewables.

- **Evaluate the appropriateness of off-shore wind in New Jersey as an electricity generation resource.** New Jersey policymakers should give serious attention to determining policies addressing the appropriateness of developing, permitting, and siting of offshore wind, a potentially large Class 1 resource in terms of a resource and economic development. While there are numerous competing arguments to encourage or discourage the development of off-shore wind, it is important that state policy address these issues on a timely basis. A timely determination may be particularly important if New Jersey decides to adopt policies to attract off-shore wind and the supporting labor force, whose economic benefits may be substantial. If New Jersey determines to encourage the development of this resource, the State's economic development policies should be integrated with its wind policies.
- **Coordinate and integrate various policies to maximize their impact.** The BPU should take specific efforts to ensure that New Jersey's policies on PVs, critical infrastructure, distributed generation, and siting are aligned. For example, if New Jersey policy seeks to maximize the reliability value of the state's investment in PVs, they should be located on the transmission and distribution system (T&D) in load pockets to reduce these costs. Similarly, PVs should be designed to function safely and reliably when grid power is not available and if possible, be located at critical facilities.
- **Strengthen the link between New Jersey's renewable energy and economic development programs by proactive initiatives to attract the location of manufacturing and maintenance facilities within the state.** The transformation of the existing electric power system requires new technologies and manufacturing and maintenance facilities, which provides an opportunity for New Jersey to attract new and expanding businesses. New Jersey is in competition with other states

for these facilities and therefore should launch proactive initiatives to realize the potential benefits to the state's economy. The BPU's efforts should be coordinated with the state's Economic Development Authority.

- **Assess whether New Jersey should develop the research and modeling capability to quantify the state specific health and environmental benefits of its renewable energy policies and to provide insights into its effectiveness.** Illustrative calculations using generic environmental externality adders indicate that in the year 2020 several hundred million dollars in environmental damage may be avoided by implementing a 20% RPS. The New Jersey BPU should lead an interagency group that includes the Department of Environmental Protection and the Department of Health to make this assessment.

III. Recommendations that Require New Jersey to Coordinate with Others

Since New Jersey is part of regional, national, and international markets for electricity, emissions, and fuels, the effectiveness of its RPS policies also depends upon the policies of other states and the federal government. To obtain the full benefits from its RPS policy, New Jersey needs to actively participate and lead a coalition of states to ensure the following:

- **Monitor the adoption of RPS policies by other states in the region, especially regarding the cost and availability of renewable resources.** Although this report assumes a static REC market in the mid-Atlantic and Northeast regions, adoption of RPS by other states will unquestionably affect the availability and costs of renewables.
- **Establish the link between reductions in emissions due to the RPS and corresponding changes in environmental policies to capture those reductions.** Existing cap-and-trade emission allowance policies for sulfur dioxide and nitrogen oxide act in combination with a RPS so that the RPS may not alone result in reduced levels of these emissions but will lower the price of emission allowances. In the case of SO_x and NO_x , obtain specific reductions in the caps on these emissions due to its RPS. If regional or national emission caps on CO_2 , CO, and particulates (PM) are established, then lower their caps over time to reflect the reductions of these emissions due to the RPS. Establishing these links should also accelerate having renewables replace existing oil and coal facilities that have worse emission rates than natural gas facilities.
- **Continue New Jersey's leadership role to facilitate trading the essential elements of RECs in the mid-Atlantic and Northeast regions to enhance liquidity in order to finance renewable projects within these regions.** RPS definitions do and are likely to continue to differ across state boundaries, which result in different types of RECs. Policies that account for these differences and permit trading the common elements of RECs across states are needed to finance renewable development.
- **Work with PJM to ensure that the wholesale market and reliability rules applicable to renewables reflect the value these resources provide.** For example, the determination of capacity contribution of PV and wind technologies is an important issue. In addition, as the percentage of renewables increase within PJM, this may raise some important reliability, transmission planning, and market issues.

Increasing the percentage of renewable resources that generate electricity for electricity consumption by New Jersey consumers reduces environmental emissions, aids in the state's economic development, provides some downward pressure on natural gas prices, and allows customers with PVs to have access to electricity when the grid is unavailable. These benefits come at a cost of a small increase in the price of electricity but have only a negligible impact to the New Jersey economy.

How to implement the proposed 20% RPS is as important as whether to do so, and there are many steps, some requiring coordination with other states and the federal government, that New Jersey policymakers should consider in order to maximize the value of the proposed 20% RPS, if they were to adopt such a policy.

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Appendix A: Supplement to the Economic Assessment

SUPPLEMENT TO THE ECONOMIC ASSESSMENT

There are many ways that increasing the size of New Jersey's renewable energy portfolio can affect the environment and the economy. One partial equilibrium outcome is that the unadjusted price of electrical energy will increase. This is because conventional sources have been selected by the market economy since they are perceived to be less expensive than alternative fuels, at least given the current electricity generation set in place. On the other hand, a point of enhancing the renewable portfolio is to meet environmental objectives. Thus, the price increase of electricity is a prime component of the costs for meeting those of objectives. Such impacts are discussed elsewhere in detail using an economic forecast as the modeling vehicle. But there are other potential costs and benefits of converting to electricity generation based on renewable energy sources. Indeed, while the relative costs of expanding investment in a renewable portfolio is embedded in average electricity prices, the relative benefits of the investments are not counted.

The benefits of a specific investment are typically measured in terms of the jobs and wealth that are created in the wake of that investment. Increases in these measures occur due to the augmentation of generation technology production, the installation of the plant, and the operation and maintenance of the facility. Naturally, to identify the changes of these measures due to conversion to a new technology, the benefits that would accrue to the new technology must be compared to the set of benefits that would have accrued had no policy change taken place. We assumed here, as throughout this report, that natural gas would serve as the prime conventional energy resource and that gas turbines would be used to generate electricity. Moreover, we presumed that conventional sources constructed in the future would be located out of state.

The net investment benefits of two renewable fuel technologies—landfill biogas and biomass—were acknowledged as negligible. Electricity produced from landfill biogas was perceived at best to maintain its current low-level production and perhaps even decline through the end of the study period. Gasified and fluidized-bed biomass technologies were recognized to be insufficiently price-competitive by 2020 (Navigant, 2004, p. 235): hence, they are supposed to make insignificant into New Jersey's energy supplies through 2020. While direct biomass conversion is presently limited, the technology being used is, in any case, largely conventional. Moreover, while policies could be developed to induce the location of such facilities in New Jersey, those same policies could serve to subsidize their eventual conversion to conventional power facilities. Hence, the economic benefits of both biomass technologies were not investigated.

The two remaining renewable energy sources for which economic benefits are considered are (1) solar photovoltaics and (2) wind. The remainder of this section discusses how their economic benefits were estimated.

In order to estimate the benefits to the New Jersey economy of these two technologies, the engineering costs of the manufacture and installation of the two

technologies had to be identified. Those for solar photovoltaics are displayed in Table A.1. It was presumed that the cost structure for photovoltaics systems connected with commercial, industrial, and civic/institutional buildings would not be much different. Engineering cost estimates for a wind park are shown in Table A.2.

Table A.1: Cost Breakdown of a Typical 8 KW Residential Solar Photovoltaic Array

Material/Service	Cost in 2004 Dollars
Marketing and Sales	3,200
Engineering and Design	480
Post-installation Servicing	960
General and Administrative	2,400
Modules	29,280
Inverters	7,680
Mounting	3,360
Miscellaneous Material	1,920
Array Labor	5,360
Electrician Labor	3,680
Total	58,320

Source: Lyle Rawlings, Advanced Solar Products, Hopewell, New Jersey.

Table A.2: An Example Cost Analysis for a 60 MW Wind Park

Item	Cost in 2004 Dollars
40 1.5 MW turbines	46,640,000
Site preparation & grid connection	9,148,000
Interest and contingencies	3,514,000
Project development & feasibility study	965,000
Engineering	611,000
Total	60,878,000

Source: Masters (2004, Table 6.8, p. 373).

With only minor modification to align the materials, services and other items so they were aligned with model sectors, the data in Tables A.1 and A.2 were entered into the R/Econ I-O model for the State of New Jersey.

For each of the two technologies, two different R/Econ I-O model runs were undertaken. In the case of PV, we first assumed that the technology was only installed in New Jersey. Prior experience has revealed that on average 90 percent of all contracting

work in New Jersey is performed by contractors who live in the state. Hence, we applied this proportion in the present study. Table A.3 shows a summary of the results on the economic and tax impacts on New Jersey of 8 Mw of PV installation in the State. Note that the direct output effects (the first column of II.1 in Table A.3) are not 1,000 times the total spending in Table A.1. This is because the PV units are assumed to be produced in New Jersey only in so much as they would normally meet state-based PV demand. Needless to say, a substantial portion of spending on solar photovoltaic units currently tends to take place outside of the State. Indeed, based on the results shown in Table A.3, only about a third (\$19 million) of the \$58 million initial investment is assumed to be spent on goods and services within the State under current circumstances, and more than half of this is for construction services required to install the arrays in place within the State. Moreover, looking at the bottom of Table A.3, it is clear that \$1 million of 2002 dollars spent on installed PV will yield 3.8 full-time-equivalent jobs, about \$280,000 in total state wealth (gross state product), \$210,000 in earnings to state workers, and a total of \$19,000 split between state and local tax revenues. Generally speaking, these are less-than-stellar returns to a state-based infrastructure investment. On the other hand, the average annual earnings per job are estimated to be \$54,700, 15% above the average for the state (\$47,420 in 2002). Of course, environmental savings of PV compared to conventional fuels are not included in these calculations.

The second run for PV assumes a policy that invokes a mandate that all PV installed in New Jersey is also produced in the State. Hence, not only were 90 percent of contractors New Jerseyans, but all design, marketing, sales, and production aspects of the PV installed in New Jersey buildings also are presumed to take place in the State. Table A.4 summarizes the economic impacts on the State of bring this policy into play. In general, the effects of this New Jersey-only technology policy yields an economic impact that is nearly two and a half times larger than if no such policy was put in place. In this case, each million spent on PV installations will produce about 8.7 jobs, \$700,000 in wealth, \$516,000 in earnings, and \$55,000 in state and local taxes. While these impacts are low in terms of jobs, they comport well otherwise with other state-based infrastructure investments. Moreover, the earnings per jobs are estimated to be even higher than in the base case—at \$59,600, 10% higher...or 25% above the state's annual average in 2002.

In the case of wind power, the basic assumption was that wind power demand would be met from sources outside of New Jersey but within the PJM market. Hence, increasing wind's share of the RPS was understood to have negligible economic impacts upon the state without prompting the industry with incentives. The other alternative is to apply policies that not only induce the installation of wind-based electric power plants within the State in the form of offshore turbines, but also to induce the production of the turbines and towers themselves. The economic impacts of this recourse are displayed in Table A.5. In general, the economic impacts per \$1 million of installing and producing wind power are just slightly lower than those for PV produced in the State: 7.6 jobs, \$636,000 in wealth, \$462,000 in earnings, and \$53,000 in state and local taxes. Due to its larger yield in manufacturing jobs, it impacts include jobs with average annual earnings of \$60,600, slightly above those for State-based PV.

**Table A.3: Economic and Tax Impacts on New Jersey
of Installing 8 MW of Residential Photovoltaic Systems
(Year 2000 Dollars)**

	Economic Component			
	Output (000\$)	Employment (jobs)	Income (000\$)	Gross State Product (000\$)
I. TOTAL EFFECTS (Direct and Indirect/Induced)*				
1. Agriculture	24.9	0	2.6	4.9
2. Agri. Serv., Forestry, & Fish	11.6	0	5.7	9.6
3. Mining	6.5	0	2.3	4.3
4. Construction	10,243.8	106	6,504.4	8,482.6
5. Manufacturing	5,553.9	25	1,467.9	1,776.0
6. Transport. & Public Utilities	1,034.3	4	263.8	415.0
7. Wholesale	693.2	4	281.9	297.7
8. Retail Trade	1,550.1	25	581.2	907.0
9. Finance, Ins., & Real Estate	1,664.8	9	554.6	1,139.8
10. Services	6,726.6	49	2,451.4	3,338.4
Private Subtotal	27,509.7	222	12,115.9	16,375.3
Public				
11. Government	58.5	0	17.9	28.5
Total Effects (Private and Public)	27,568.3	222	12,133.8	16,403.7
II. DISTRIBUTION OF EFFECTS/MULTIPLIER				
1. Direct Effects	19,113.5	156	9,326.1	12,246.5
2. Indirect and Induced Effects	8,454.8	66	2,807.6	4,157.3
3. Total Effects	27,568.3	222	12,133.8	16,403.7
4. Multipliers (3/1)	1.442	1.427	1.301	1.339
III. COMPOSITION OF GROSS STATE PRODUCT				
1. Wages--Net of Taxes				11,230.0
2. Taxes				2,204.7
a. Local				276.3
b. State				252.9
c. Federal				1,675.5
General				374.6
Social Security				1,300.9
3. Profits, dividends, rents, and other				2,969.0
4. Total Gross State Product (1+2+3)				16,403.7
IV. TAX ACCOUNTS				
	Business	Household	Total	
1. Income --Net of Taxes	11,230.0	0.0		
2. Taxes	2,204.7	2,462.1	4,666.8	
a. Local	276.3	315.6	591.9	
b. State	252.9	276.3	529.2	
c. Federal	1,675.5	1,870.2	3,545.7	
General	374.6	1,870.2	2,244.8	
Social Security	1,300.9	0.0	1,300.9	
EFFECTS PER MILLION DOLLARS OF INITIAL EXPENDITURE				
Employment (Jobs)				3.8
Income				208,055.4
State Taxes				9,073.4
Local Taxes				10,149.5
Gross State Product				281,270.8
INITIAL EXPENDITURE IN DOLLARS				58,320,000.0
Note: Detail may not sum to totals due to rounding.				
*Terms:				
Direct Effects --the proportion of direct spending on goods and services produced in the specified region.				
Indirect Effects--the value of goods and services needed to support the provision of those direct economic effects.				
Induced Effects--the value of goods and services needed by households that provide the direct and indirect labor.				

**Table A.4: Economic and Tax Impacts on New Jersey
of Installing 8 MW of Residential Photovoltaic Systems,
Assuming All Production Takes Place in the State
(Year 2000 Dollars)**

		Economic Component			
		Output (000\$)	Employment (jobs)	Income (000\$)	Gross State Product (000\$)
I. TOTAL EFFECTS (Direct and Indirect/Induced)*					
	1. Agriculture	60.4	0	6.4	11.7
	2. Agri. Serv., Forestry, & Fish	47.9	1	24.7	40.2
	3. Mining	9.4	0	3.2	6.1
	4. Construction	11,173.3	108	6,632.8	8,792.5
	5. Manufacturing	46,388.8	193	14,318.2	18,542.0
	6. Transport. & Public Utilities	2,960.6	11	724.8	1,180.8
	7. Wholesale	2,645.1	15	1,075.6	1,136.1
	8. Retail Trade	3,767.7	61	1,408.6	2,192.6
	9. Finance, Ins., & Real Estate	4,375.3	22	1,427.9	3,005.2
	10. Services	11,454.2	93	4,432.3	5,691.5
	Private Subtotal	82,882.7	504	30,054.6	40,598.7
	Public				
	11. Government	180.1	1	55.3	89.3
	Total Effects (Private and Public)	83,062.9	505	30,109.9	40,687.9
II. DISTRIBUTION OF EFFECTS/MULTIPLIER					
	1. Direct Effects	58,320.0	319	21,894.6	28,733.6
	2. Indirect and Induced Effects	24,742.9	186	8,215.3	11,954.3
	3. Total Effects	83,062.9	505	30,109.9	40,687.9
	4. Multipliers (3/1)	1.424	1.582	1.375	1.416
III. COMPOSITION OF GROSS STATE PRODUCT					
	1. Wages--Net of Taxes				27,650.1
	2. Taxes				5,688.1
	a. Local				943.7
	b. State				787.3
	c. Federal				3,957.1
	General				728.9
	Social Security				3,228.3
	3. Profits, dividends, rents, and other				7,349.8
	4. Total Gross State Product (1+2+3)				40,687.9
IV. TAX ACCOUNTS					
		Business	Household	Total	
	1. Income --Net of Taxes	27,650.1	0.0		
	2. Taxes	5,688.1	6,109.6		11,797.6
	a. Local	943.7	783.1		1,726.8
	b. State	787.3	685.6		1,472.9
	c. Federal	3,957.1	4,640.8		8,598.0
	General	728.9	4,640.8		5,369.7
	Social Security	3,228.3	0.0		3,228.3
EFFECTS PER MILLION DOLLARS OF INITIAL EXPENDITURE					
	Employment (Jobs)				8.7
	Income				516,287.6
	State Taxes				25,254.9
	Local Taxes				29,609.3
	Gross State Product				697,666.5
INITIAL EXPENDITURE IN DOLLARS					58,320,000.0

Note: Detail may not sum to totals due to rounding.

*Terms:

Direct Effects --the proportion of direct spending on goods and services produced in the specified region.

Indirect Effects--the value of goods and services needed to support the provision of those direct economic effects.

Induced Effects--the value of goods and services needed by households that provide the direct and indirect labor.

**Table A.5: Economic and Tax Impacts on New Jersey
of Installing 60 MW Off-shore Wind Park,
Assuming All Production Takes Place in the State
(Year 2000 Dollars)**

	Economic Component			
	Output (000\$)	Employment (jobs)	Income (000\$)	Gross State Product (000\$)
I. TOTAL EFFECTS (Direct and Indirect/Induced)*				
1. Agriculture	60.2	0	6.2	11.5
2. Agri. Serv., Forestry, & Fish	46.7	1	24.1	39.2
3. Mining	108.3	1	37.3	71.1
4. Construction	6,601.3	40	3,066.3	4,340.5
5. Manufacturing	56,821.3	240	16,624.7	21,836.4
6. Transport. & Public Utilities	7,360.5	21	1,457.5	2,812.5
7. Wholesale	2,999.4	17	1,219.7	1,288.3
8. Retail Trade	3,704.1	60	1,382.6	2,150.1
9. Finance, Ins., & Real Estate	4,240.8	22	1,420.2	2,854.4
10. Services	6,510.1	62	2,838.8	3,233.3
Private Subtotal	88,452.7	463	28,077.5	38,637.1
Public				
11. Government	218.5	1	66.6	105.3
Total Effects (Private and Public)	88,671.2	464	28,144.0	38,742.4
II. DISTRIBUTION OF EFFECTS/MULTIPLIER				
1. Direct Effects	60,878.0	262	18,727.9	25,558.5
2. Indirect and Induced Effects	27,793.2	203	9,416.1	13,183.9
3. Total Effects	88,671.2	464	28,144.0	38,742.4
4. Multipliers (3/1)	1.457	1.775	1.503	1.516
III. COMPOSITION OF GROSS STATE PRODUCT				
1. Wages--Net of Taxes				25,637.1
2. Taxes				5,524.4
a. Local				1,033.9
b. State				841.2
c. Federal				3,649.3
General				631.8
Social Security				3,017.5
3. Profits, dividends, rents, and other				7,580.9
4. Total Gross State Product (1+2+3)				38,742.4
IV. TAX ACCOUNTS				
	Business	Household	Total	
1. Income --Net of Taxes	25,637.1	0.0		
2. Taxes	5,524.4	5,710.7	11,235.1	
a. Local	1,033.9	732.0	1,765.9	
b. State	841.2	640.8	1,482.1	
c. Federal	3,649.3	4,337.8	7,987.1	
General	631.8	4,337.8	4,969.6	
Social Security	3,017.5	0.0	3,017.5	
EFFECTS PER MILLION DOLLARS OF INITIAL EXPENDITURE				
Employment (Jobs)				7.6
Income				462,302.2
State Taxes				24,344.8
Local Taxes				29,007.5
Gross State Product				636,394.0
INITIAL EXPENDITURE IN DOLLARS				60,878,000

Note: Detail may not sum to totals due to rounding.

*Terms:

Direct Effects --the proportion of direct spending on goods and services produced in the specified region.

Indirect Effects--the value of goods and services needed to support the provision of those direct economic effects.

Induced Effects--the value of goods and services needed by households that provide the direct and indirect labor.

This part of Appendix A includes the BASELINE forecast used in this study as well as 4 other scenarios discussed in Chapter 2.

Table A.6: R/ECON BPU BASELINE FORECAST

	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	Growth 2004 - 2020
Electric Utilities: Prices, Usage, Taxes, etc.																		
Electric Price per Kilowatt Hour																		
Residential	\$0.100	\$0.100	\$0.100	\$0.100	\$0.101	\$0.102	\$0.102	\$0.103	\$0.103	\$0.104	\$0.105	\$0.105	\$0.106	\$0.106	\$0.107	\$0.108	\$0.108	8.0%
Other	\$0.111	\$0.110	\$0.110	\$0.110	\$0.110	\$0.111	\$0.111	\$0.112	\$0.112	\$0.113	\$0.113	\$0.114	\$0.115	\$0.115	\$0.116	\$0.117	\$0.117	5.4%
	\$0.093	\$0.095	\$0.095	\$0.095	\$0.097	\$0.097	\$0.098	\$0.098	\$0.099	\$0.099	\$0.100	\$0.100	\$0.101	\$0.102	\$0.102	\$0.103	\$0.104	11.8%
Electricity Usage in 1000 Megawatt Hours																		
Residential	66,537	73,644	75,482	76,615	77,620	78,629	79,635	80,621	81,636	82,679	83,794	84,952	86,064	87,151	88,242	89,368	90,539	36.1%
Other	26,367	26,538	26,946	27,405	27,864	28,315	28,758	29,195	29,632	30,073	30,522	30,982	31,450	31,923	32,397	32,872	33,350	26.5%
	40,170	47,106	48,536	49,210	49,756	50,314	50,877	51,425	52,004	52,607	53,271	53,970	54,614	55,228	55,845	56,496	57,189	42.4%
Electric Revenues (\$ Billions)																		
Residential	\$6.7	\$7.4	\$7.6	\$7.7	\$7.9	\$8.0	\$8.2	\$8.3	\$8.5	\$8.6	\$8.8	\$8.9	\$9.1	\$9.3	\$9.5	\$9.7	\$9.8	47.8%
Other	\$2.9	\$3.0	\$3.0	\$3.0	\$3.1	\$3.1	\$3.2	\$3.3	\$3.3	\$3.4	\$3.4	\$3.5	\$3.6	\$3.7	\$3.8	\$3.8	\$3.9	33.3%
	\$3.7	\$4.5	\$4.6	\$4.7	\$4.8	\$4.9	\$5.0	\$5.0	\$5.1	\$5.2	\$5.3	\$5.4	\$5.5	\$5.6	\$5.7	\$5.8	\$5.9	59.2%
Energy Taxes (\$ Millions)																		
Energy Taxes - TEFA	\$1,108.3	\$932.6	\$897.7	\$856.7	\$869.4	\$881.4	\$893.8	\$907.2	\$922.2	\$938.4	\$955.6	\$973.5	\$991.3	\$1,008.5	\$1,025.4	\$1,042.5	\$1,060.3	-4.3%
Sales	\$858.8	\$822.6	\$842.7	\$856.7	\$869.4	\$881.4	\$893.8	\$907.2	\$922.2	\$938.4	\$955.6	\$973.5	\$991.3	\$1,008.5	\$1,025.4	\$1,042.5	\$1,060.3	23.5%
Corporate Business	\$697.1	\$630.1	\$639.4	\$647.8	\$658.5	\$667.4	\$675.7	\$684.4	\$693.6	\$703.0	\$713.2	\$723.7	\$734.1	\$744.5	\$755.1	\$766.1	\$777.6	11.5%
Transitional Facility Assessment	\$161.7	\$192.6	\$203.3	\$208.9	\$210.9	\$214.0	\$218.1	\$222.9	\$228.6	\$235.4	\$242.5	\$249.8	\$257.2	\$264.0	\$270.3	\$276.4	\$282.7	74.8%
	\$249.5	\$110.0	\$55.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	
Gross State Product for Utilities (\$ Billions)	\$8.04	\$8.16	\$8.40	\$8.52	\$8.56	\$8.63	\$8.72	\$8.82	\$8.95	\$9.09	\$9.25	\$9.41	\$9.57	\$9.72	\$9.86	\$9.99	\$10.12	26.0%
Gross State Product (\$Billions 2000=100)	\$354.57	\$364.10	\$373.83	\$383.97	\$393.62	\$403.29	\$414.16	\$426.09	\$438.96	\$453.03	\$468.20	\$484.17	\$500.86	\$518.30	\$536.48	\$555.63	\$575.88	62.4%
Employment at Utilities (Thousands)	15.4	15.3	15.4	15.4	15.4	15.3	15.3	15.3	15.3	15.3	15.2	15.2	15.2	15.2	15.2	15.2	15.2	-1.2%
Electric Utilities	8.9	8.8	8.7	8.6	8.6	8.5	8.5	8.4	8.4	8.4	8.4	8.3	8.3	8.3	8.3	8.3	8.3	-6.4%
Other	6.4	6.6	6.7	6.8	6.8	6.8	6.9	6.9	6.9	6.9	6.9	6.9	6.9	6.9	6.9	6.9	6.9	7.7%
NonAgricultural Employment (Thousands)	4030.2	4079.7	4133.8	4183.6	4227.1	4270.3	4317.4	4363.2	4411.9	4463.4	4521.9	4576.9	4628.1	4677.9	4730.5	4784.8	4845.0	20.2%
NJ Consumer Price Index (1982=100) ¹	196.6	199.4	201.8	205.3	209.5	213.9	218.7	224.2	229.7	235.3	241.1	247.1	253.2	259.4	265.8	272.4	279.1	41.9%
Renewable Portfolio Standard	3.25	3.40	4.13	5.12	6.21	6.50	6.50	6.50	6.50	6.50	6.50	6.50	6.50	6.50	6.50	6.50	6.50	100.0%
Class 1: Photovoltaics, et al.	0.75	0.90	1.63	2.62	3.71	4.00	4.00	4.00	4.00	4.00	4.00	4.00	4.00	4.00	4.00	4.00	4.00	433.3%
Class 2:Hydroelectric and Waste-to-Energy	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	0.0%

¹ The NJ CPI is a population-weighted average of the CPIs for NY-NJ-CT and PA-NJ.

**Table A.7: R/ECON BPU STRAIGHTLINE RPS INCREASE
2009 to 20% in 2020**

	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
Electric Utilities: Prices, Usage, Taxes, etc.																	
Electric Price per KiloWatt Hour																	
Residential	\$0.100	\$0.100	\$0.100	\$0.100	\$0.101	\$0.103	\$0.105	\$0.107	\$0.110	\$0.113	\$0.116	\$0.119	\$0.122	\$0.125	\$0.128	\$0.131	\$0.134
Other	\$0.111	\$0.110	\$0.110	\$0.110	\$0.110	\$0.111	\$0.112	\$0.114	\$0.117	\$0.119	\$0.122	\$0.125	\$0.127	\$0.130	\$0.133	\$0.136	\$0.138
Electricity Usage in 1000 Megawatt Hours																	
Residential	\$0.093	\$0.095	\$0.095	\$0.095	\$0.097	\$0.098	\$0.100	\$0.103	\$0.106	\$0.109	\$0.112	\$0.115	\$0.119	\$0.122	\$0.125	\$0.129	\$0.131
Other	66,536	73,643	75,480	76,614	77,619	78,552	79,516	80,484	81,489	82,527	83,639	84,797	85,909	86,996	88,089	89,216	90,444
Electric Revenues (\$ Billions)																	
Residential	\$6.7	\$7.4	\$7.6	\$7.7	\$7.9	\$8.1	\$8.3	\$8.6	\$9.0	\$9.3	\$9.7	\$10.1	\$10.5	\$10.9	\$11.3	\$11.7	\$12.1
Other	\$2.9	\$2.9	\$3.0	\$3.0	\$3.1	\$3.1	\$3.2	\$3.3	\$3.5	\$3.6	\$3.7	\$3.9	\$4.0	\$4.1	\$4.3	\$4.5	\$4.6
Energy Taxes (\$ Millions)																	
Total - TEPA	\$1,078.8	\$932.6	\$897.6	\$856.6	\$869.5	\$884.8	\$904.6	\$927.3	\$952.9	\$980.8	\$1,010.6	\$1,041.8	\$1,073.4	\$1,105.0	\$1,136.7	\$1,169.0	\$1,198.5
Sales	\$858.8	\$822.6	\$842.6	\$856.6	\$869.5	\$884.8	\$904.6	\$927.3	\$952.9	\$980.8	\$1,010.6	\$1,041.8	\$1,073.4	\$1,105.0	\$1,136.7	\$1,169.0	\$1,198.5
Corporate Business	\$697.1	\$630.1	\$639.4	\$647.8	\$658.5	\$670.8	\$686.5	\$704.6	\$724.5	\$745.8	\$768.5	\$792.4	\$816.6	\$841.3	\$866.7	\$892.9	\$916.1
Transitional Facility Assessment	\$161.7	\$192.5	\$203.2	\$208.8	\$211.0	\$214.0	\$218.1	\$222.7	\$228.4	\$235.0	\$242.1	\$249.4	\$256.8	\$263.6	\$270.0	\$276.1	\$282.4
Gross State Product for Utilities (\$ Billions)	\$220.0	\$110.0	\$55.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0
Gross State Product (\$ Billions 2000=100)	\$8.04	\$8.16	\$8.39	\$8.52	\$8.56	\$8.63	\$8.72	\$8.82	\$8.94	\$9.09	\$9.24	\$9.40	\$9.56	\$9.71	\$9.85	\$9.98	\$10.12
Employment at Utilities (Thousands)																	
Electric Utilities	\$354.56	\$364.07	\$373.80	\$383.94	\$393.60	\$403.28	\$414.13	\$426.02	\$438.82	\$452.81	\$467.91	\$483.79	\$500.38	\$517.73	\$535.83	\$554.89	\$575.05
Other	15.4	15.3	15.4	15.4	15.4	15.3	15.3	15.3	15.3	15.3	15.2	15.2	15.2	15.2	15.2	15.2	15.2
Nonagricultural Employment (Thousands)	4030.1	4079.6	4133.7	4183.5	4227.0	4270.2	4317.3	4363.0	4411.5	4462.8	4521.2	4576.0	4627.0	4676.6	4729.0	4783.0	4843.0
NJ Consumer Price Index (1982-24=100) ¹	196.8	199.4	201.8	205.3	209.5	214.0	218.9	224.5	230.2	235.9	241.7	247.8	254.0	260.3	266.8	273.5	280.2
Reasonable Portfolio Standard	3.25	3.40	4.13	5.12	6.21	7.22	8.44	9.67	10.90	12.13	13.35	14.58	15.81	17.03	18.26	19.49	20.00
Class 1: Photovoltaics, et al	0.75	0.90	1.63	2.62	3.71	4.72	5.94	7.17	8.40	9.63	10.85	12.08	13.31	14.53	15.76	16.99	17.50
Class 2: Hydroelectric and Waste-to-Energy	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50
1. The NJ CPI is a population-weighted average of the CPIs for NY-NJ-CT and PA-NJ.																	

Table A.8: R/ECON BPU BASELINE WITH HIGH PPI ENERGY

	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	Growth 2004 - 2020
Electric Utilities: Prices, Usage, Taxes, etc.																		
Electric Price per Kilowatt Hour																		
Residential	\$0.100	\$0.101	\$0.101	\$0.102	\$0.103	\$0.104	\$0.105	\$0.106	\$0.107	\$0.107	\$0.108	\$0.109	\$0.110	\$0.111	\$0.112	\$0.112	\$0.113	13.0%
Other	\$0.093	\$0.095	\$0.096	\$0.097	\$0.099	\$0.100	\$0.100	\$0.101	\$0.102	\$0.103	\$0.103	\$0.104	\$0.105	\$0.106	\$0.107	\$0.108	\$0.109	17.2%
Electricity Usage in 1000 Megawatt Hours																		
Residential	66,537	73,628	75,441	76,576	77,582	78,600	79,613	80,608	81,625	82,671	83,786	84,943	86,055	87,141	88,233	89,358	90,529	36.1%
Other	26,367	26,537	26,940	27,398	27,856	28,309	28,753	29,192	29,630	30,071	30,521	30,981	31,449	31,921	32,396	32,871	33,348	26.5%
	40,170	47,091	48,501	49,178	49,726	50,291	50,860	51,416	51,995	52,600	53,265	53,963	54,606	55,220	55,837	56,487	57,181	42.3%
Electric Revenues (\$ Billions)																		
Residential	\$6.7	\$7.4	\$7.6	\$7.8	\$8.0	\$8.2	\$8.4	\$8.6	\$8.7	\$8.9	\$9.1	\$9.3	\$9.5	\$9.7	\$9.9	\$10.1	\$10.3	55.1%
Other	\$3.7	\$4.5	\$4.7	\$4.8	\$4.9	\$5.0	\$5.1	\$5.2	\$5.3	\$5.4	\$5.5	\$5.6	\$5.7	\$5.9	\$6.0	\$6.1	\$6.2	40.2%
																		66.8%
Energy Taxes (\$ Millions)																		
Energy Taxes - TEFA	\$1,108.3	\$933.0	\$899.7	\$860.7	\$876.6	\$891.7	\$906.7	\$922.1	\$938.5	\$955.9	\$974.2	\$993.3	\$1,012.3	\$1,030.9	\$1,049.3	\$1,068.0	\$1,087.4	-1.9%
Sales	\$858.8	\$823.0	\$844.7	\$860.7	\$876.6	\$891.7	\$906.7	\$922.1	\$938.5	\$955.9	\$974.2	\$993.3	\$1,012.3	\$1,030.9	\$1,049.3	\$1,068.0	\$1,087.4	26.6%
Corporate Business	\$697.1	\$630.6	\$642.6	\$653.9	\$667.5	\$679.0	\$689.3	\$699.5	\$709.9	\$720.5	\$731.8	\$743.6	\$755.3	\$767.1	\$779.2	\$791.8	\$805.0	15.5%
Transitional Facility Assessment	\$161.7	\$192.4	\$202.1	\$206.8	\$209.1	\$212.7	\$217.4	\$222.6	\$228.6	\$235.3	\$242.4	\$249.7	\$257.0	\$263.7	\$270.1	\$276.2	\$282.5	74.6%
	\$249.5	\$110.0	\$55.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	
Gross State Product for Utilities (\$ Billions)	\$8.04	\$8.16	\$8.37	\$8.47	\$8.52	\$8.60	\$8.70	\$8.82	\$8.95	\$9.09	\$9.25	\$9.41	\$9.56	\$9.71	\$9.85	\$9.98	\$10.12	25.9%
Gross State Product (\$Billions 2006=100)	\$354.57	\$364.09	\$373.80	\$383.92	\$393.57	\$403.25	\$414.14	\$426.08	\$438.96	\$453.02	\$468.20	\$484.16	\$500.85	\$518.29	\$536.47	\$555.62	\$575.88	62.4%
Employment at Utilities (Thousands)																		
Electric Utilities	15.4	15.3	15.4	15.4	15.4	15.3	15.3	15.3	15.3	15.3	15.2	15.2	15.2	15.2	15.2	15.2	15.2	-1.2%
Other	8.9	8.8	8.7	8.6	8.6	8.5	8.5	8.4	8.4	8.4	8.4	8.3	8.3	8.3	8.3	8.3	8.3	-6.4%
	6.4	6.6	6.7	6.8	6.8	6.8	6.9	6.9	6.9	6.9	6.9	6.9	6.9	6.9	6.9	6.9	6.9	7.7%
NonAgricultural Employment (Thousands)	4030.2	4079.7	4133.8	4183.6	4227.1	4270.2	4317.4	4363.2	4411.9	4463.4	4521.9	4576.9	4628.1	4677.9	4730.5	4784.8	4845.0	20.2%
NJ Consumer Price Index (1982=100) ¹	196.6	199.4	201.9	205.4	209.5	213.9	218.7	224.2	229.8	235.3	241.1	247.1	253.2	259.4	265.8	272.4	279.1	41.9%
Renewable Portfolio Standard	3.25	3.40	4.13	5.12	6.21	6.50	6.50	6.50	6.50	6.50	6.50	6.50	6.50	6.50	6.50	6.50	6.50	100.0%
Class 1: Photovoltaics, et al.	0.75	0.90	1.63	2.62	3.71	4.00	4.00	4.00	4.00	4.00	4.00	4.00	4.00	4.00	4.00	4.00	4.00	433.3%
Class 2: Hydroelectric and Waste-to-Energy	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	0.0%

¹ The NJ CPI is a population-weighted average of the CPIs for NY-NJ-CT and PA-NJ.

**Table A.9: R/ECON BPU STRAIGHTLINE RPS INCREASE 2009 to 20% IN 2020
OUT-OF-STATE MANUFACTURING OF RENEWABLE TECHNOLOGIES**

Electric Utilities: Prices, Usage, Taxes, etc.	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
Electric Price per Kilowatt Hour	\$0.100	\$0.100	\$0.100	\$0.100	\$0.101	\$0.102	\$0.102	\$0.104	\$0.104	\$0.106	\$0.107	\$0.108	\$0.108	\$0.109	\$0.110	\$0.111	\$0.112
Residential	\$0.111	\$0.111	\$0.110	\$0.110	\$0.110	\$0.111	\$0.111	\$0.112	\$0.113	\$0.114	\$0.114	\$0.115	\$0.116	\$0.117	\$0.118	\$0.119	\$0.120
Other	\$0.093	\$0.095	\$0.095	\$0.095	\$0.097	\$0.097	\$0.098	\$0.099	\$0.100	\$0.102	\$0.103	\$0.103	\$0.104	\$0.104	\$0.106	\$0.107	\$0.108
Electric Usage in 100 Megawatt Hours																	
Residential	66,537	73,645	75,484	76,618	77,625	78,611	79,622	80,566	81,617	82,592	83,781	84,939	86,059	87,162	88,214	89,358	90,535
Other	26,367	26,538	26,946	27,405	27,864	28,308	28,753	29,193	29,625	30,068	30,520	30,974	31,444	31,920	32,390	32,867	33,347
	40,170	47,107	48,538	49,213	49,761	50,302	50,869	51,372	51,993	52,524	53,261	53,964	54,615	55,242	55,825	56,490	57,188
Electric Revenues (\$ Billions)																	
Residential	\$6.7	\$7.4	\$7.6	\$7.7	\$7.9	\$8.0	\$8.2	\$8.4	\$8.5	\$8.8	\$9.0	\$9.1	\$9.3	\$9.5	\$9.7	\$10.0	\$10.2
Other	\$2.9	\$2.9	\$3.0	\$3.0	\$3.1	\$3.1	\$3.2	\$3.3	\$3.3	\$3.4	\$3.5	\$3.6	\$3.6	\$3.7	\$3.8	\$3.9	\$4.0
	\$3.7	\$4.5	\$4.6	\$4.7	\$4.8	\$4.9	\$5.0	\$5.1	\$5.2	\$5.4	\$5.5	\$5.6	\$5.7	\$5.7	\$5.9	\$6.0	\$6.2
Energy Taxes (\$ Millions)																	
Total - TEPA	\$1,108.3	\$932.6	\$897.7	\$856.7	\$869.5	\$881.9	\$894.9	\$910.9	\$927.2	\$949.0	\$966.8	\$986.2	\$1,004.5	\$1,021.0	\$1,041.0	\$1,059.8	\$1,081.3
Sales	\$658.8	\$822.6	\$842.7	\$856.7	\$869.5	\$881.9	\$894.9	\$910.9	\$927.2	\$949.0	\$966.8	\$986.2	\$1,004.5	\$1,021.0	\$1,041.0	\$1,059.8	\$1,081.3
Corporate Business	\$697.1	\$630.1	\$639.4	\$647.8	\$658.5	\$667.9	\$676.8	\$688.1	\$698.7	\$713.8	\$724.4	\$736.4	\$747.4	\$757.0	\$770.7	\$783.3	\$798.6
Transitional Facility Assessment	\$161.7	\$192.6	\$203.3	\$208.9	\$211.0	\$214.0	\$218.1	\$222.9	\$228.5	\$235.3	\$242.4	\$249.8	\$257.1	\$264.0	\$270.3	\$278.5	\$282.7
	\$249.5	\$110.0	\$55.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0
Gross State Product for Utilities (\$ Billions 2000=100)	\$8.13	\$8.26	\$8.50	\$8.62	\$8.67	\$8.73	\$8.82	\$8.93	\$9.05	\$9.20	\$9.36	\$9.52	\$9.68	\$9.83	\$9.97	\$10.11	\$10.25
Gross State Product (\$Billions 2000=100)	\$380.81	\$391.04	\$401.50	\$412.39	\$422.76	\$433.15	\$444.83	\$457.63	\$471.43	\$486.52	\$502.81	\$519.94	\$537.86	\$556.60	\$576.13	\$596.71	\$618.47
Employment at Utilities (Thousands)	15.4	15.3	15.4	15.4	15.4	15.3	15.3	15.3	15.3	15.3	15.2	15.2	15.2	15.2	15.2	15.2	15.2
Electric Utilities	8.9	8.8	8.7	8.6	8.6	8.5	8.5	8.4	8.4	8.4	8.4	8.3	8.3	8.3	8.3	8.3	8.3
Other	6.4	6.6	6.7	6.8	6.8	6.8	6.9	6.9	6.9	6.9	6.9	6.9	6.9	6.9	6.9	6.9	6.9
NonAgricultural Employment (Thousands)	4030.2	4079.8	4134.0	4183.9	4227.6	4271.4	4317.9	4363.5	4412.5	4463.9	4522.5	4577.5	4629.1	4678.9	4731.7	4786.3	4846.8
NJ Consumer Price Index (1982=100) ¹	196.6	199.4	201.8	205.3	209.5	213.9	218.8	224.3	229.9	235.4	241.2	247.2	253.3	259.5	265.9	272.5	279.2
Renewable Portfolio Standard	3.25	3.40	4.13	5.12	6.21	7.22	8.44	9.67	10.90	12.13	13.35	14.58	15.81	17.03	18.26	19.49	20.00
Class 1: Photovoltaics, et al	0.75	0.90	1.63	2.62	3.71	4.72	5.94	7.17	8.40	9.63	10.85	12.08	13.31	14.53	15.76	16.99	17.50
Class 2: Hydroelectric and Waste-to-Energy	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50

¹ The NJ CPI is a population-weighted average of the CPIs for NY-NJ-CT and PA-NJ.

**Table A.10: R/ECON BPU STRAIGHTLINE RPS INCREASE 2009 to 20% IN 2020 WITH
MANUFACTURING AND MAINTENANCE OF NEW TECHNOLOGIES INSTATE**

R/ECON BPU STRAIGHTLINE RPS INCREASE 2009 to 20% IN 2020 WITH MANUFACTURING AND MAINTENANCE OF NEW TECHNOLOGIES INSTATE	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
Electric Utilities: Prices, Usage, Taxes, etc.																	
Electric Price per KiloWatt Hour	\$0.100	\$0.100	\$0.100	\$0.100	\$0.101	\$0.102	\$0.102	\$0.104	\$0.104	\$0.106	\$0.107	\$0.108	\$0.108	\$0.109	\$0.110	\$0.111	\$0.112
Residential	\$0.111	\$0.111	\$0.110	\$0.110	\$0.110	\$0.111	\$0.111	\$0.112	\$0.113	\$0.114	\$0.114	\$0.115	\$0.116	\$0.117	\$0.118	\$0.119	\$0.120
Other	\$0.093	\$0.095	\$0.095	\$0.095	\$0.097	\$0.097	\$0.098	\$0.099	\$0.100	\$0.102	\$0.103	\$0.103	\$0.104	\$0.104	\$0.106	\$0.107	\$0.108
Electricity Usage in 1000 Megawatt Hours	66,537	73,648	75,494	76,629	77,651	78,664	79,679	80,611	81,674	82,658	83,852	85,019	86,153	87,263	88,325	89,486	90,679
Residential	26,367	26,538	26,946	27,406	27,865	28,310	28,756	29,198	29,630	30,073	30,527	30,981	31,452	31,929	32,399	32,878	33,359
Other	40,170	47,110	48,548	49,223	49,786	50,354	50,923	51,413	52,045	52,585	53,326	54,038	54,701	55,335	55,926	56,608	57,320
Electric Revenues (\$ Billions)	\$6.7	\$7.4	\$7.6	\$7.7	\$7.9	\$8.0	\$8.2	\$8.4	\$8.6	\$8.8	\$9.0	\$9.1	\$9.3	\$9.5	\$9.8	\$10.0	\$10.2
Residential	\$2.9	\$2.9	\$3.0	\$3.0	\$3.1	\$3.1	\$3.2	\$3.3	\$3.3	\$3.4	\$3.5	\$3.6	\$3.6	\$3.7	\$3.8	\$3.9	\$4.0
Other	\$3.7	\$4.5	\$4.6	\$4.7	\$4.8	\$4.9	\$5.0	\$5.1	\$5.2	\$5.4	\$5.5	\$5.6	\$5.7	\$5.8	\$5.9	\$6.1	\$6.2
Energy Taxes (\$ Millions)	\$1,108.3	\$932.7	\$897.8	\$856.8	\$869.7	\$882.3	\$895.4	\$911.4	\$927.7	\$949.6	\$967.3	\$986.8	\$1,005.2	\$1,021.7	\$1,041.9	\$1,060.7	\$1,082.4
Total - TEPA	\$658.8	\$622.7	\$642.8	\$656.8	\$669.7	\$682.3	\$695.4	\$711.4	\$727.7	\$749.6	\$767.3	\$786.8	\$1,005.2	\$1,021.7	\$1,041.9	\$1,060.7	\$1,082.4
Sales	\$697.1	\$630.1	\$639.5	\$647.9	\$658.7	\$668.3	\$677.2	\$688.4	\$699.0	\$714.2	\$724.9	\$737.0	\$748.0	\$757.7	\$771.4	\$784.2	\$799.5
Corporate Business	\$161.7	\$192.6	\$203.3	\$208.9	\$211.0	\$214.1	\$218.2	\$223.0	\$228.7	\$235.4	\$242.5	\$249.8	\$257.2	\$264.1	\$270.5	\$276.6	\$282.9
Transitional Facility Assessment	\$249.5	\$110.0	\$55.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0	\$0.0
Gross State Product for Utilities (\$ Billions 2005=100)	\$8.13	\$8.26	\$8.50	\$8.62	\$8.67	\$8.73	\$8.83	\$8.93	\$9.06	\$9.20	\$9.36	\$9.52	\$9.69	\$9.84	\$9.98	\$10.11	\$10.25
Gross State Product (\$Billions 2005=100)	\$380.81	\$391.06	\$401.57	\$412.50	\$422.93	\$433.40	\$445.16	\$458.02	\$471.89	\$487.03	\$503.37	\$520.57	\$538.55	\$557.34	\$576.95	\$597.61	\$619.4542
Employment at Utilities (Thousands)	15.4	15.3	15.4	15.4	15.4	15.4	15.3	15.3	15.3	15.3	15.3	15.2	15.2	15.2	15.2	15.2	15.2
Electric Utilities	8.9	8.8	8.7	8.6	8.6	8.5	8.5	8.4	8.4	8.4	8.4	8.3	8.3	8.3	8.3	8.3	8.3
Other	6.4	6.6	6.7	6.8	6.8	6.8	6.9	6.9	6.9	6.9	6.9	6.9	6.9	6.9	6.9	6.9	6.9
NonAgricultural Employment (Thousands)	4030.2	4080.5	4134.8	4185.1	4231.0	4276.4	4321.4	4367.2	4417.3	4469.1	4528.1	4584.2	4636.6	4686.7	4740.7	4796.7	4855.2
NU Consumer Price Index (1982-24=100) ¹	196.6	199.4	201.8	205.3	209.5	213.9	218.8	224.3	229.9	235.4	241.2	247.2	253.3	259.5	265.9	272.5	279.2
Renewable Portfolio Standard	3.25	3.40	4.13	5.12	6.21	7.22	8.44	9.67	10.90	12.13	13.35	14.58	15.81	17.03	18.26	19.49	20.00
Class 1 Photovoltaics, et al.	0.75	0.90	1.63	2.62	3.71	4.72	5.94	7.17	8.40	9.63	10.85	12.08	13.31	14.53	15.76	16.99	17.50
Class 2 Hydroelectric and Waste-to-Energy	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50	2.50

¹ The NU CPI is a population-weighted average of the CPIs for NY-NJ-CT and PA-NJ.

Appendix B: Supplement to the Environmental Externality Analysis

SUPPLEMENT TO THE ENVIRONMENTAL EXTERNALITY ANALYSIS

1. INTRODUCTION

The purpose of this appendix is to expand upon the environmental externality discussion in Chapter 3.

This appendix uses a benefit transfer approach. This method consists of conducting a thorough literature review of recent, North American applicable, peer-reviewed studies. The key question in following this approach is how closely other studies resemble the situation in New Jersey based on timeliness, methodology, and other factors. Ideally, primary research based on the characteristics of New Jersey is preferred to a benefit transfer approach, but the types of benefits are too numerous and their phenomenon too complex to pursue primary research given the financial and time constraints of this project.

The general steps taken to estimate non-market benefits of air pollution abatement are as follows:

1. Identifying Benefits
2. Quantifying Benefits
3. Monetizing Benefits

The first step involves describing a qualitative relationship between changes in pollutant emissions and ambient concentrations, and subsequently between ambient concentrations and environmental effects.

Identifying the benefits of air pollution abatement is equivalent to identifying the damages that are reduced or avoided. These damages fall into three broad categories (adopted from Freeman, 1993):

1. Direct damages to humans
2. Indirect damages to humans through ecosystems
3. Indirect damages to humans through nonliving systems

Direct damages to humans include health damages, as well as aesthetic damages such as unpleasant odor, noise or poor visibility. Indirect damages to humans through ecosystems consist of productivity damages in the form of crop reduction, and damages to forests and commercial fisheries; recreation damages (lakes, rivers, etc.), and intrinsic or nonuse damages. The latter are damages to ecological resources that are not motivated by people's own use of these resources. For example, people value endangered species or rare ecosystems, even though they do not have the intention to ever see or experience them. Finally, indirect damages to humans that occur through nonliving things include damages to materials and structures, such as soiling and corrosion.

The second step, quantifying benefits, involves establishing a functional relationship between environmental effects and the reduction in air pollution. For example, such

quantitative relationships can be described by dose-response or concentration-response functions. These functions describe the change in a health effect, say, asthma attacks, and the concentration of the pollutant that causes the effect. In order to calculate the number of cases that will be avoided, we need to establish a baseline exposure (number of people affected, and the level of pollution they are subjected to) and the baseline number of cases for each quantifiable health effect for each pollutant. These numbers are then contrasted with the number of cases for each quantifiable health effect with the regulation (RPS, in our case), to calculate the number of cases avoided as a result of the RPS.

The third step, monetizing or valuing the benefits, is typically specific to the environmental effect under consideration. In what follows, we describe separately the most common valuation methods used in the literature for each effect.

As part of the process of evaluating the evidence presented by this body of literature, each study must be evaluated as to the soundness of the data, the analytic techniques and the conclusions drawn by the authors.

Although the approach of this report is to determine whether these non-market benefits can be monetized in the context of NJ RPS policymaking, there are other ways of accounting for these benefits in policymaking without monetization. One method is tradeoff analysis, and another involves a deliberation process. If tradeoff analysis were to be applied here, then each of the non-market benefits would be quantified separately and not combined into a dollar value. Tradeoffs between different impacts, such as costs and illnesses due to sulfur dioxide emissions, would be evaluated. Such an analysis can also be informed by stakeholders' values pertaining to the relative importance of different impacts have. If we used a deliberative process, for example one mandated by law, various stakeholders would provide input. An outcome is considered successful if the process is successfully completed.

2. CRITERIA FOR BENEFIT TRANSFER

Instead of conducting primary research, valuation studies often apply the benefit transfer method. Benefit transfer entails using monetary values estimated in existing empirical studies to assess the value of a quantified effect in a different study.

When is Benefit Transfer applicable?

1. When existing studies value similar effects.
2. When the context of the existing studies is highly similar.
3. When the existing studies are of high quality.

There are no universally accepted criteria for benefit transfer, but in most cases the defensibility of the benefit estimates depends on the quality of the existing study. Hence one criterion for benefit transfer is to select studies that were published in peer-reviewed journals, and have been viewed highly by the professional community.

Benefit transfer may increase the inherent uncertainty surrounding environmental benefits estimates, but for practical reasons (e.g. cost and time), benefit transfer is a necessary component of policy analysis. In most situations, the question is not whether to conduct benefit transfer, but how to improve benefit transfer to make it more reliable.

When one uses benefit transfer, there are three sources of variation between the original study location/situation and the one to which the benefits estimates are transferred to that one has to consider. The first is individual variation (x), the second is commodity variation (y), and the third is other variation (z).

Consider the following hypothetical situation. Suppose that the state of New Jersey is planning to upgrade a state park, and one proposal is to provide a swimming beach to the lake. The benefit subject to valuation is recreational swimming at the lake. Suppose researchers have identified studies from other parts of the country that value similar benefits. The estimates of recreational swimming benefits may not accurately measure the benefit of recreational swimming in New Jersey for three main reasons: 1. The preferences of New Jersey residents may differ from the preferences of the participants of the selected studies. This is the individual variation (x) between the participants of the original study, and the population to which benefit transfer is applied. 2. The characteristics of the benefit may also differ. For example, the benefit derived from the availability of a clean beach for swimming may be different in Alaska as it is in New Jersey. This is called the commodity variation (y). 3. The first two sources of variation imply that the value of recreational swimming in New Jersey is different from the context of the original study, while the third source of variation (z) implies that the value was incorrectly estimated. Each type of variation has a fixed and a random component (see table below).

	Individual Variation x	Commodity Variation y	Other Variation z
Fixed Component	μ_I	μ_C	μ_O
Random Component	ε_I	ε_C	ε_O

The fixed component of each variation includes variables that can be measured and observed by the researchers, while random variations are not observable. Hence, we can make benefit transfer more reliable by minimizing the fixed component of each variation. Individual variation may be reduced, if one is able to estimate how the value of recreational swimming varies with demographic and socioeconomic variables, such as age, income, etc. Using the data on these observable variables, one may adjust the estimates for the policy context. Similarly, if one identifies the relationship between the benefits provided by the different commodities, then the second type of variation is reduced. For example, one might conduct a small calibration survey for the policy site. The results from the original study could then be weighted by the respondents' attitude/experience values for the policy site to calibrate the transfer. Selecting high-quality studies for the policy context minimizes the third type of variation. The main criteria in study selection should include statistical tests of the explanatory power and robustness of the models used in estimation. The calibration of the original estimates may reduce the variation between the policy context and the original study, and hence enhance the defensibility of benefit transfer.

3. DIRECT BENEFITS TO HUMANS – HEALTH BENEFITS

Linkages between air pollution and health effects are subtle and often difficult to establish. However, the available literature provides strong evidence that air pollution has adverse effects on human health. In general, the information that is needed for the estimation of health benefits is similar to the input needed for estimating non-health related benefits of air pollution abatement. The first step is to estimate the change in emissions resulting from the regulatory action. The next step involves air quality modeling to estimate the relationship between changes in emissions and changes in ambient pollutant concentrations. Third, one must determine the population and risk, in order to be able to estimate the health effects. Fourth, the health effects must be monetized to conduct a cost-benefit analysis.

3.1 IDENTIFYING HEALTH BENEFITS

Health benefits resulting from reduced air pollution can be grouped into two broad categories: mortality benefits and morbidity benefits. Mortality benefits are avoided deaths from diseases caused by air pollution. Morbidity benefits refer to avoided cases of non-fatal health effects. Air pollutants that have been linked to adverse health effects include particulate matter (PM), ozone (O₃), carbon monoxide (CO), sulfur dioxide (SO₂), and nitrogen dioxide (NO₂). Until the mid-1980's it was believed that ambient pollutant concentrations did not have adverse health effects (Katsouyanni, 2003). Over the last 15 years, a large body of epidemiological literature has been devoted to the study of adverse health effects occurring at moderate and low pollutant concentrations. There is now sufficient evidence to support the hypothesis that both chronic and acute health effects can occur at ambient pollution levels. Current research focuses on the consequence of acute and chronic air pollution exposure for excess cardiovascular and respiratory morbidity and mortality.

Types of health studies

There are two main types of methods to establish a dose-response relationship for a health effect:

1. Epidemiological studies (cohort studies, case control studies, occupational epidemiology studies, cross-sectional studies)
2. Toxicological studies (mechanistic studies, animal studies, human studies)

Epidemiological studies attempt to establish a quantitative relationship between health effect and air pollution using a sample drawn from a large population. The four most commonly used types of epidemiological studies are cohort studies, case control studies, occupational epidemiology studies, and cross-sectional studies.

Cohort studies follow a group of healthy people with different levels of exposure and assess the impact of exposure on their health over time. The term cohort in epidemiology refers to a collection of people that share some common characteristics, such as age or ethnicity. The two common types of cohort studies, prospective and retrospective, both start by identifying and enrolling subjects based upon the presence or absence of exposure, without knowing whether the exposure resulted in any adverse impact. In a typical prospective cohort study, individuals in a cohort are followed forward in time, for a sufficiently long period, to track the appearance of a disease and disability. On the contrary, in retrospective cohort studies first the cohorts are selected and then the exposure histories of the participants are collected and studied. Cohort studies are used because in this setting the issue of temporality is controlled, since the exposure precedes the disease process. Besides producing more reliable estimates, prospective cohort studies also allow us to study how multiple risk factors determine the onset and history of one or more diseases.

Case control studies identify a group of individuals with a certain health condition, as well as a group of subjects without the health effect (control group), and try to answer the question why some people got ill, while the those in the control group did not. Case control studies are less time-intensive and expensive than cohort studies, but they are also subject to a greater estimation bias.

Occupational epidemiology studies people working in particular jobs as subjects. Workers in certain occupations often have a higher exposure to a pollutant than the general population, and hence it may be easier to identify a causal relationship between exposure to a pollutant and the health effect. Occupational epidemiology studies may not be appropriate for benefit transfer because their study populations may not represent the general populations in terms of risk.

Cross-sectional studies analyze the relationship between a group's (e.g. a metropolitan area) health status and exposure status simultaneously. This study design does not allow for changes in variables over time, and hence it may fail to uncover the true relationship between exposure and a health outcome.

In toxicology, mechanistic studies examine how and why various disease processes occur in response to toxicant exposures, and help establish a relationship between dose or exposure and response. Animal studies also provide precise information about the adverse response to a substance because the studies are controlled in a laboratory, and animals are subjected to a wide range of exposures. One of the advantages of animal studies is the researcher's ability to extrapolate from the high doses in animal studies down to the low doses often experienced in human exposure scenarios. Finally, human studies can be used to extrapolate the response of humans at low doses to higher doses.

Long-term (chronic) health effects of air pollution have been evaluated by a large number of cross-sectional and a few prospective-cohort studies (Dockery et al., 1993; Pope et al., 1995; HEI, 2000; Pope et al., 2002). Prospective cohort study is the desirable design because exposure precedes the health outcome, which is a necessary condition for

establishing a causal relationship between exposure and the health outcome. Moreover, this study design is less subject to bias because exposure is evaluated before the health status is known, and also more accurate data may be collected. Künzli and Tager (2000) argue that cross-sectional and prospective cohort studies address different aspects of the association between air pollution and mortality. Cross-sectional studies are only capable of capturing mortality effects triggered by air pollution exposures that occurred shortly before death, while prospective cohort studies capture all air pollution-related mortality effects.

Health benefits due to particulate matter reductions

Adverse health effects of exposure to particles have been described in numerous epidemiological studies. Health endpoints include all-cause and cause-specific mortality and hospital admissions. Studies conducted in the United States and in other countries have reported associations between changes in PM and changes in mortality and morbidity, particularly among subgroups of people with respiratory or cardiovascular diseases. However, the exact mechanisms by which PM influences human health are not well understood. Earlier literature focused on PM greater than 10 μ m in diameter, while in the last decade the attention of researchers turned to fine particles such as PM_{2.5}. Recent research indicates that ultrafine particles (UF) less than 0.1 μ m in diameter may play an important role in the induction of toxic effects. Currently, however, data on UF exposure and health effects are still limited.

Although the importance of long-term exposure to PM has been emphasized, most of the attention in the literature has been devoted to short-term health effects. Two prominent prospective-cohort studies of mortality effects of PM are Dockery et al. (1993) and Pope et al. (1995). Both of these are prospective cohort studies. Unlike earlier studies, Dockery et al. (1993) estimate the effect of air pollution on mortality while controlling for individual risk factors. Pope et al. (1995) study the association between air pollution and mortality using data from a large cohort drawn from many study areas. Pope et al. (2002) is a continuation of the Pope et al. (1995), while HEI (2000) study is a reanalysis of the original Pope et al. (1995) data. Information on data, methodology, and the results of these four studies are summarized in Table 3.1.

Short-term, or acute effects of PM are well established for morbidity endpoints such as, hospital admissions for respiratory and cardiovascular conditions. There is also evidence of acute effects on respiratory function, lower respiratory symptoms, and increased medication use by asthmatics (Katsouyanni, 2003). There are fewer studies available on the long-term, or chronic, health effects of PM pollution. A few studies have linked an increase in chronic bronchitis occurrence to an increase in ambient PM concentration (Abbey et al., 1993; Schwartz, 1993; Abbey et al., 1995). Tables 3.6-3.9, at the end of this chapter, summarize the available studies that assessed morbidity effects resulting in chronic and minor illness, as well as hospital admissions. We selected studies that were conducted in the United States and Canada, and found a positive association between the health effect and air pollution.

Table 3.1. Prospective Cohort Studies of Mortality Due to Particulate Matter Pollution				
Study	Study Location and Population	Study Period	Results	
Dockery et al. (1993)	8111 adults in six U.S. cities	14-to-16-year mortality follow-up	Air pollution was positively associated with death from lung cancer and cardiopulmonary disease but not with death from other causes considered together. Mortality was most strongly associated with air pollution with fine particulates, including sulfates.	
Pope et al. (1995)	552,138 adults in 151 U.S. metropolitan areas	Ambient pollution data: 1980 Enrollment: 1982 Deaths through: 1989	PM pollution associated with cardiopulmonary and lung cancer mortality but not with mortality due to other causes. Increased mortality is associated with sulfate and fine particulate air pollution at levels commonly found in U.S. cities.	
HEI (2000)	552,138 adults in 151 U.S. metropolitan areas	Ambient pollution data: 1980 Enrollment: 1982 Deaths through: 1989	The original results of Dockery et al. (1993) and Pope et al. (1995) were successfully replicated and validated. In alternative models, estimated mortality effects increased in the subgroup of subjects with less than high school education. When sulfur dioxide was included in models with fine particles or sulfate, the associations between these pollutants (fine particles and sulfate) and mortality diminished.	
Pope et al. (2002)	319,000-590,000 adults in 51-102 U.S. metropolitan areas, depending on the PM measure and study period	Enrollment: 1982 Deaths through: 1998	Fine particle and sulfur oxide pollution were associated with all-cause death, lung cancer and cardiopulmonary mortality. Each 10 $\mu\text{g}/\text{m}^3$ increase in fine PM pollution was associated with approximately 4%, 6%, and 8% increase in the risk of all-cause death, cardiopulmonary mortality, and lung cancer mortality, respectively.	

Health benefits due to ozone reductions

Ozone is formed by a chemical reaction from its precursor pollutants (volatile organic compounds (VOCs) and nitrogen oxides (NO, NO₂), and nitrous oxide (N₂O)) in the presence of heat and sunlight. Ozone concentration is the highest in the summer when the weather is hot and sunny with relatively light winds. Electric power plants are among the main sources of precursors pollutant emissions.

Health problems are caused by tropospheric, or ground-level, ozone. Ozone is associated with a variety of adverse health effects ranging from minor symptoms to hospital admissions and chronic illness. Some studies have found a link between ozone and mortality, however there is significant uncertainty about the relationship between mortality and high ozone concentrations, partly because of the possible confounding effect of other pollutants such as particulate matter. Table 3.2 below summarizes the most common adverse health effects associated with ozone.

Table 3.2. Likely Ozone-related Adverse Health Effects	
Adverse Health Effect	Comment
Respiratory Hospital Admissions	A large number of studies have linked ozone to hospital admissions for pneumonia, chronic obstructive pulmonary disease (COPD), asthma and other respiratory ailments.
Cardiovascular Hospital Admissions	There is a link between high ozone and dysrhythmias (abnormal heartbeat patterns).
Total Respiratory ER Visits	Studies have also found a link between high ozone and emergency room visits which do not result in actual hospital admissions.
Minor Symptoms	Short-term exposure to ozone has been linked to a variety of symptoms, including cough, sore throat and head cold.
Asthma Attacks	Ozone has specifically been linked to incidence of asthma attacks and may be linked to the development of chronic asthma.
Shortness of breath	Ozone associated with shortness of breath in asthmatics and non-asthmatics.
Source: Abt Associates (1999)	

Mechanistic studies of ozone yield a sufficient evidence for a biologic plausibility of respiratory-related morbidity and mortality. For a review of mechanistic studies of ozone see Levy et al. (2001). There is evidence from human and animal exposure studied that long-term exposure to ozone may cause a sustained decrement in lung function. There are well-documented molecular mechanisms for acute respiratory effects of ozone, but the

evidence for chronic respiratory effects is limited. There is, however, increasing evidence that high ozone can result in the development of chronic diseases. For example, McConnell (2002) provided the first evidence suggesting that tropospheric ozone causes the development of childhood asthma. In high ozone-concentration cities, children who played outdoor sports were 3 to 4 times more likely to develop chronic asthma than children who did not play sports. In low ozone-concentration cities, children playing sports were no more likely to develop asthma than children who did not play sports.

Because indoor ozone concentrations are generally lower than ambient concentrations, personal exposure may not be directly related to ambient concentration. Personal exposures to ozone are influenced by air conditioning or averting behavior, such as, more time spent indoors. When applying concentration-response functions, one must determine the relationship between ambient ozone concentrations and personal exposures. An understanding of any systemic differences between the study and policy region is crucial.

The importance of personal exposures to ozone led to new recent stream of research where the analysis of health effects is stratified by some relevant personal characteristics (e.g. insurance status) or regional characteristics (e.g. prevalence of air conditioning). Two recent studies, Gwynn and Thurston (2001) and Nauenberg and Basu (1999) find that insurance status as a factor in the strength of the association between ozone and hospital admissions for asthma. Jaffe et al. (2003) adjust for insurance status in the relationship between air pollution and emergency department (ED) visits, by looking at asthma ED visits and ozone among Medicaid recipients. Finkelstein et al. (2000) find that among predictors of emergency room visits for asthma is insurance status.

Health benefits due to carbon monoxide reductions

Carbon monoxide (CO) is a colorless and odorless gas produced through incomplete combustion of carbon-based fuels. Carbon monoxide enters the bloodstream through the lungs and reduces the delivery of oxygen to the body's organs and tissues. The most vulnerable to CO are those who suffer from cardiovascular disease, particularly those with angina or peripheral vascular disease. Fetuses and young infants, children, pregnant women, individuals with obstructive pulmonary disease such as bronchitis and emphysema, smokers, and individuals spending a lot of time on the street working or doing exercise are also more susceptible to CO exposure. In health studies, high CO concentrations have been linked to hospital admissions for asthma, chronic obstructive pulmonary disease (COPD), dysrhythmias, ischemic heart disease, and congestive heart failure (CHF).

Health benefits due to sulfur dioxide reductions

Sulfur dioxide is formed when fuel, containing sulfur, such as coal and oil, is burned. The main health effects associated with exposure to high SO₂ concentrations include effects on breathing, respiratory illness, changes in pulmonary defenses, and aggravation of cardiovascular disease. The most susceptible groups are children, the elderly, asthmatics,

and people with cardiovascular and chronic lung disease (such as bronchitis and emphysema). High SO₂ levels have been linked to the following endpoints: hospital admissions for pneumonia, ischemic heart disease, and respiratory conditions; chest tightness, shortness of breath, and wheeze.

Health benefits due to nitrogen dioxide reductions

NO₂ is a suffocating, brownish gas that is formed when fuel is burned at high temperatures. Primary sources of NO₂ are motor vehicles, electric utilities and industrial boilers. Nitrogen dioxide can irritate the lungs and lower resistance to respiratory infections such as influenza. There is no clear evidence on the effect of short-term exposure to NO₂ on health, but frequent exposure may cause an increased incidence of acute respiratory illness, especially in children. NO₂ has been linked to hospital admissions for respiratory conditions, pneumonia, congestive heart failure, and ischemic heart disease. Epidemiological studies found that NO₂ has a modifying effect on PM: the increase in mortality due to PM was found to be higher in cities where long-term NO₂ concentrations were higher (Katsouyanni, 2003). In addition NO₂ may have other indirect adverse effects, as it contributes to ozone formation. Therefore the importance of NO₂ for health comes from its role as an O₃ precursor and a contributor to the formation of secondary particles.

3.2 QUANTIFYING HEALTH BENEFITS

Health benefits are typically estimated using the damage-function (DF) method the consists of the following steps involved:

1. Determining the dose-response relationship for each health effect
2. Determining baseline exposure
3. Determining the number of baseline cases for each quantifiable health effect
4. Number exposed × Baseline exposure × Dose-response relationship.
5. Determining exposure after the regulation (for each regulatory option)
6. Determining the number of cases for each quantifiable effect with the regulation
7. Determining the number of cases avoided as a result of each regulatory option

The purpose of quantification is to determine the change in the occurrence of a health effect (y) as a result of a change in pollutant concentrations (x) between the baseline and the control scenario. Such relationship between, say particulate matter (PM) concentration (Δx), and the change in the health effect (Δy) is described by dose-response or concentration-response (CR) functions. Dose-response or concentration-response (CR) functions estimate the risk (of the occurrence of a health effect) per unit of exposure to a pollutant. The two most common functional forms for the CR relationship between air pollutants and a health effect (e.g. mortality) used in the literature are log-linear and linear functions. The linear relationship can be described by the following equation:

$$y = \alpha + \beta \cdot x$$

Where α and β are the parameters to be estimated. From the above equation it follows that:

$$\Delta y = \alpha + \beta \cdot \Delta x$$

Log-linear CR functions have the following general form:

$$y = \gamma \cdot e^{\beta \cdot x}$$

Or, equivalently:

$$\log(y) = \alpha + \beta \cdot x \quad \text{where } \alpha = \log(\gamma).$$

Let y_1 denote the occurrence of the health effect under the baseline scenario, and let y_2 be a measure of the health effect under the control scenario. Then the relationship between the change in PM concentration and the health effect may be written as follows:

$$\Delta y = y_1 - y_2 = -\gamma \cdot e^{\beta \cdot PM_1} \left[\frac{e^{\beta \cdot x_1}}{e^{\beta \cdot x_2}} - 1 \right] = -y_1 \left[e^{-\beta \cdot \Delta x} - 1 \right]$$

In a sufficiently large population, some people develop a disease that can be attributed to air pollution regardless of whether they were exposed or not. Relative risk (RR) is a measure that tells us the seriousness of exposure to a known risk factor. It is defined as the risk is for those exposed relative to those who are not exposed. For example, if the risk of developing a disease in the exposed population is 5%, while in the non-exposed population it is 1%, the relative risk is 5. A high risk factor indicates strong evidence between exposure to a pollutant and the health effect.

The risk factor associated with a log-linear CR function has the following form:

$$RR_{\log\text{-linear}} = \frac{y_2}{y_1} = e^{-\beta \cdot \Delta x}$$

Epidemiological studies typically report the relative risk, rather than the CR-coefficients. The above equations may be used to calculate the coefficients from the reported RR-values.

Use of thresholds in CR functions

Typically, dose-response functions that have been estimated for health effects describe a linear no-threshold relationship. This means that every unit of exposure contributes equally to aggregate risk in a large population of people. For example, a linear no-threshold dose-response function treats the case of one person being exposed to one hundred units of the pollutant, and ten people subjected to ten units of pollution equally (simply, as 100 units of human exposure). In some cases, the use of such dose-response functions may not be appropriate. Thresholds may be incorporated into the analysis even when one uses CR-functions that were derived under the no-threshold assumptions. While the possible existence of a threshold in concentration-response relationships is an

important scientific question, there is currently no scientific basis for selecting appropriate threshold levels.

Health studies estimating CR relationship often attempt to justify their linearity assumption. For example, support for the no-threshold assumption in mortality is provided by a recent study by Vedal et al. (2003). They study the association between daily inhalable particle concentrations and daily mortality in Vancouver, British Columbia, where daily average PM₁₀ and ozone concentrations have been very low during the study period. After analyzing three years of data, they conclude that increases in low concentrations of air pollution are associated with daily mortality.

To determine the number of baseline cases, we need to identify the segment of population that is exposed, and the number of people exposed within each segment, as well as the level, duration and frequency of exposure. In order to obtain accurate estimates, we also have to control for averting behavior (that is, people with known risk may act to avoid exposure).

Quantifying Mortality Benefits

In valuation studies mortality benefits linked to particulate matter (PM) tend to dominate total monetized benefits of air pollution abatement. The relationship between mortality and ambient PM concentration is well established, while this is not the case for other pollutants. Moreover, there is some evidence that there are synergistic effects between PM and other pollutants (e.g. ozone). Therefore it is desirable to transfer estimates from studies that consider multiple pollutants as explanatory variables in regression models.

Quantifying Mortality due to Particulate Matter

In epidemiological studies of PM, typical measurable health endpoints include all-cause and cause specific mortality, as well as hospital admissions and emergency room visits. Tables 3.3 and 3.4 summarize relative risk estimates associated with PM pollution from the four available prospective cohort studies. The Pope et al. (2002) study estimates relative risk associated with a 10 $\mu\text{g}/\text{m}^3$ increase in PM_{2.5} (particulates less than 2.5 μm in diameter), while the HEI (2000) study consider a 25 $\mu\text{g}/\text{m}^3$ change. Therefore, the relative risk estimates in these tables are not comparable. At the end of this chapter, in Tables 3.13-3.45 we list dose-response functions from available studies, and for each relative risk estimate we derived an estimate for β . The value of $100 \times \beta$ can be interpreted as the percentage change in the health effect associated with a unit increase in the pollutant.

Table 3.3: Pope et al. (2002) estimates of adjusted relative risk (RR) associated with a 10 $\mu\text{g}/\text{m}^3$ change in fine particles measuring less than 2.5 μm in diameter

Cause of Mortality	Adjusted relative risk (95% confidence interval)		
	1979-1983	1999-2000	Average
All causes	1.04 (1.01-1.08)	1.06 (1.02-1.10)	1.06 (1.02-1.11)
Cardiopulmonary	1.06 (1.02-1.10)	1.08 (1.02-1.14)	1.09 (1.03-1.16)
Lung cancer	1.08 (1.01-1.16)	1.13 (1.04-1.22)	1.14 (1.04-1.23)
All other causes	1.01 (0.97-1.05)	1.01 (0.97-1.06)	1.01 (0.95-1.06)

Source: Pope et al. (2002), Table 2

Relative risk is adjusted for age, sex, race, smoking, education, body mass, alcohol consumption, occupational exposure, and diet.

Table 3.4: Relative Risks of Mortality Associated with a 24.5 $\mu\text{g}/\text{m}^3$ Increase in Fine Particles Using Alternative Measures of PM in the Reanalysis of the Pope et al. (1995) study.

Cause of Mortality	Adjusted relative risk (95% confidence interval)		
	Median PM _{2.5} Pope et al. (1995) measure	Median PM _{2.5} HEI (2000) measure	Mean PM _{2.5} Pope et al. (2000) measure
All cause	1.18 (1.09–1.26)	1.14 (1.06–1.22)	1.12 (1.06–1.19)
Cardiopulmonary	1.30 (1.17–1.44)	1.26 (1.14–1.39)	1.26 (1.16–1.38)
Lung cancer	1.00 (0.79–1.28)	1.08 (0.88–1.32)	1.08 (0.88–1.32)

Source: HEI (2000), Summary Table 4, p.21

Relative risks were calculated for a change in the pollutant of interest equal to the difference in mean concentrations between the most-polluted city and the least-polluted city. In the Pope et al. (1995) study, this difference for fine particles was 24.5 $\mu\text{g}/\text{m}^3$.

HEI (2000) tested whether the relationship between ambient concentrations/exposure and mortality was linear using the data of Pope et al. Support for both linear and nonlinear relationships was found, depending upon the analytic technique used. Both Pope et al. (1995) and Dockery et al. (1993) used the Cox proportional hazard regression model, under which hazard functions for mortality at two pollutant levels are proportional and invariant in time.

HEI (2000) evaluate the applicability of the Cox proportional hazard model, using flexible concentration-response models, for the data used in Dockery et al. (1993) and Pope et al. (1995). The small number of study locations in Dockery et al. (1993) afforded only a limited opportunity to define the shape of the CR-function. No evidence was found against the linearity of the relationship for PM. For sulfate particles, however, there was some evidence of departures from linearity at both low and high sulfate concentrations. A similar analysis of the Pope et al. (1995) data yielded some evidence of departure from linearity for both PM and sulfate particles. Overall, however, the Cox proportional hazards assumption did not appear inappropriate. Finally, Pope et al. (2002) conclude that within the range of pollution observed in the study, the CR relationship appears to be monotonic and nearly linear.

Quantifying Mortality due to Ozone

There is considerable epidemiologic evidence concerning the relationship between ambient ozone concentrations and human mortality risks. Because ozone contributes to acute (short-term) health effects, the association between daily ozone concentrations and daily mortality is of primary interest to researchers. Table 3.5 below summarizes the findings of a number of studies that found a statistically significant relationship between daily mortality and daily ozone concentrations. These studies show mixed findings as to whether there is a statistically significant association between daily ozone concentrations and daily mortality in each of the study areas.

Table 3.5 Studies for daily mortality and ambient ozone				
Study	Study Location/Duration	Co-pollutants in the model	O₃ concentration measure (ppb)	Relative risk and 95% confidence interval for a 25 ppb increase in O₃
Ito and Thurston (1996)	Cook County, Illinois 1985-1990	PM ₁₀	Average same day and previous day 1-hr maxima	1.016 (1.004-1.029)
Kinney et al. (1995)	Los Angeles County 1985-1990	PM ₁₀	Daily 1-hour maximum	1.000 (0.089-1.010)
Verhoeff et al. (1996)	Amsterdam, Netherlands 1986-1992	PM ₁₀ Black smoke	Daily 1-hour maximum (2-day lag)	with PM ₁₀ 1.024 (0.974 -1.078) with black smoke 1.014 (0.984 - 1.046)
Anderson et al. (1997)	London, England 1987-1992	Black smoke	8-hour average and daily 1-hour max (1-day lag)	1.029 (1.015 - 1.042)
Kwon et al. (2001) ²	Seoul, South Korea 1994-1998	TSP, SO ₂ , NO ₂ , CO, O ₃	8-hour average	1.01 (1.002-1.017)
Goldberg et al. (2001)	Montreal, Quebec 1984-1993	O ₃ , SO ₂ , NO ₂ , CO, PM ₁₀ , PM _{2.5}	Daily average, 3-day running mean	1.033 (1.017-1.05)
Hong et al. (2002) ³	Seoul, South Korea 1991-1997	TSP, SO ₂ , Lagged NO ₂ , CO, O ₃	8-hour average	1.06 (1.02-1.10)
Moolgavkar et al. (1995)	Philadelphia 1973-1988	TSP, SO ₂	Daily average	1.015 (1.004-1.026)
Samet et al. (1997)	Philadelphia 1973-1988	TSP, SO ₂ , NO ₂ Lagged CO	2-day average	1.024 (1.008-1.039)
Notes: 1. O ₃ was measured in µg/m ³ . To convert to ppb, ozone concentrations in µg/m ³ were divided by 1.96. In general, the conversion factor depends on the temperature in the study area. 2. The effect of air pollution on daily mortality of patients with congestive heart failure was studied. 3. Only ischemic stroke mortality was included in the study.				

Quantifying Mortality due to Carbon Monoxide

A number of studies examined the relationship between daily mortality and concentrations of CO. Cardiovascular mortality was found strongly associated with CO concentrations. The table below summarizes the results and characteristics of several studies.

Table 3.6
Studies for Carbon Monoxide and Mortality

Study	Location, Study Period and Population	Pollutants	Endpoints	Main Findings	Comment
Burnett et al. (1998)	Toronto, Canada 1980-1984 All ages	CO, NO ₂ , SO ₂ , O ₃ , SO ₄ , TSP, COH, PM ₁₀ , PM _{2.5}	Non-accidental mortality	Significant effect found in all two-pollutant models. Controlling for CO, significant effect found for SO ₄ , TSP, COH, PM ₁₀ and PM _{2.5}	Association with cardiac-related mortality is stronger, but CO is also significantly related to non-cardiac mortality. PM ₁₀ and PM _{2.5} estimated from SO ₄ , TSP and COH
Kinney et al. (1995)	Los Angeles County 1985-1990 All ages	CO, O ₃ , PM ₁₀	Non-accidental mortality	In single pollutant model, CO is significant, and PM ₁₀ and O ₃ are marginally significant. In the model with CO and PM ₁₀ , both CO and PM ₁₀ are significant.	Magnitude of single-pollutant CO relationship drops modestly with the inclusion of PM ₁₀ .
Saldiva et al. (1995)	São Paulo, Brazil 1990-91 Elderly (65+ years)	CO, O ₃ , PM ₁₀ , SO ₂ , NO _x	Mortality from natural causes	CO significant in single-pollutant model, not significant in any multi-pollutant model.	
Touloumi et al. (1996)	Athens, Greece 1987-1991 All ages	CO, SO ₂ , black smoke	Total mortality	CO, SO ₂ , and black smoke significant in single pollutant models	Deaths during one month summertime heat wave were excluded from the analysis

Table 3.6
Studies for Carbon Monoxide and Mortality (continued)

Study	Location, Study Period and Population	Pollutants	Endpoints	Main Findings	Comment
Mar et al. (2000)	Phoenix, Arizona 1995-1997	PM _{2.5} , PM ₁₀ , SO ₂ , NO ₂ , CO	Cardiovascular mortality	Cardiovascular mortality was significantly associated with CO, NO ₂ , SO ₂ , PM _{2.5} , PM ₁₀ , PM _{CF} and elemental carbon.	Both combustion-related pollutants and secondary aerosols (sulfates) were associated with cardiovascular mortality.
Hong et al. (2002)	Seoul, South Korea 1991-1997	TSP, SO ₂ , Lagged NO ₂ , CO, O ₃	Ischemic Stroke Mortality (ICD9:431.434; ICD10: I61, I63)	Significantly increased relative risks were found for same-day TSP and SO ₂ , for NO ₂ and CO with a 1-day lag, and for O ₃ with a 3-day lag for each interquartile range increase in the pollutant concentration.	

Quantifying Morbidity Benefits

Quantifying morbidity benefits is more difficult for chronic (long-term) conditions than for acute (short-term) health effects, because it requires data on exposure over a long period of time. The most frequently used endpoints in the epidemiologic literature are hospital admissions for various respiratory and cardiovascular illnesses, emergency department visits, chronic diseases (e.g. chronic bronchitis), and minor health effects, such as upper respiratory symptoms (URS), lower respiratory symptoms (LRS), asthma attacks, shortness of breath, work loss days, minor restricted activity days, etc.

Table 3.7 below briefly describes the available studies that found an association between chronic illness and air pollution. Table 3.8 lists the available studies for minor illness, and Tables 3.9 and 3.10 summarize studies of hospital admissions for respiratory and cardiovascular causes, respectively.

Table 3.7
Studies for Chronic Illness and Air Pollution

<i>Study</i>	<i>Location, Study Period and Population</i>	<i>Pollutants</i>	<i>Endpoints</i>	<i>Main Findings</i>	<i>Comment</i>
Portney and Mullahy (1990)	Nationwide Sample from the 1979 National Health Interview Survey 1,318 persons age 17-93	O ₃ , TSP	Sinusitis, hay fever, AOD	Controlling for TSP, O ₃ significantly related to the initiation (or exacerbation) of sinusitis and hay fever; no effect on AOD. TSP not significantly related to any endpoint, although it is marginally significant for AOD.	
Schwartz (1993)	Nationwide sample from the National Health and Nutrition Examination Survey 1974-75 6,138 individuals ages 30-74	TSP, SO ₂	Chronic bronchitis, asthma, shortness of breath (dyspnea), respiratory illness	TSP significantly related to the prevalence of chronic bronchitis, and marginally significant for respiratory illness. No effect on asthma or dyspnea.	Respiratory illness defined as a significant condition, coded by an examining physician as ICD8 code (460-519)
Xu et al. (1993)	Beijing, China, Survey conducted August-September 1986 1,576 never-smokers	TSP, SO ₂	Chronic bronchitis, asthma	Chronic bronchitis significantly higher in the community with the highest TSP level. TSP not linked to the prevalence of asthma.	
Zemp et al. (1999)	Eight sites in Switzerland 1991 9,651 individuals ages 18-60	TSP, PM ₁₀ , NO ₂ , O ₃	Chronic phlegm, chronic cough, breathlessness, asthma, dyspnea in exertion	Single-pollutant models: PM ₁₀ and NO ₂ significantly associated with chronic cough or phlegm, breathlessness and dyspnea. Similar though less significant associations found for TSP. No significant effect found for O ₃ .	

Table 3.7
Studies for Chronic Illness and Air Pollution (continued)

<i>Study</i>	<i>Location, Study Period and Population</i>	<i>Pollutants</i>	<i>Endpoints</i>	<i>Main Findings</i>	<i>Comment</i>
Abbey et al. (1993)	California initial survey: 1977 final survey: 1987 3,914 Seventh Day Adventists	TSP, O ₃ , SO ₂	AOD, chronic bronchitis, asthma	TSP linked to new cases of AOD and chronic bronchitis, but not to asthma or the severity of asthma. O ₃ not linked to the incidence of new cases of any endpoint, but O ₃ was linked to the severity of asthma. No effect found for SO ₂ .	Emphysema, chronic bronchitis, and asthma comprise AOD
Abbey et al. (1995)	California initial survey: 1977 final survey: 1987 1,868 Seventh Day Adventists	PM _{2.5}	AOD, chronic bronchitis, asthma	PM _{2.5} related to new cases of chronic bronchitis, but not to new cases of AOD or asthma	PM _{2.5} estimated from visibility data
Chapman et al. (1985)	4 Utah communities 1976 5,623 young adults	SO ₂ , SO ₄ , NO ₃ , TSP	Persistent cough and phlegm	Persistent cough and phlegm is higher in the community with higher SO ₂ , SO ₄ and TSP concentrations	
McDonnell et al. (1999)	California initial survey: 1997 final survey: 1992 3,091 Seventh Day Adventists	O ₃ , PM ₁₀ , SO ₂ , SO ₄ , NO ₂	Asthma	Single-pollutant models: O ₃ significantly linked to asthma cases in males, but not females; other pollutants not significantly linked to new asthma cases in males or females. Two-pollutant models estimated for ozone with another pollutant; little impact found on size of ozone coefficient	Average pollution level from 1973-1992 used. Prior to 1987, PM ₁₀ from TSP.

Table 3.8
Studies for Minor Illness

<i>Endpoints</i>	<i>Study Population</i>	<i>Study</i>	<i>Pollutants Used in Final Model</i>
Acute bronchitis	Ages 8-12	Dockery et al. (1996)	PM _{2.5}
Upper respiratory symptoms (URS)	Ages 9-11	Pope et al. (1991)	PM ₁₀
Lower respiratory symptoms (LRS)	Ages 7-14	Schwartz et al. (1994)	PM _{2.5}
Respiratory illness	Ages 6-7	Hasselblad et al. (1992)	NO ₂
Any of 19 respiratory symptoms	Ages 18-65	Krupnick et al. (1990)	O ₃ , PM ₁₀
Moderate or worse asthma	All ages (asthmatics)	Ostro et al. (1991)	PM _{2.5}
Asthma attacks	All ages (asthmatics)	Whittemore and Korn (1980)	O ₃ , PM ₁₀
Chest tightness, shortness of breath, or wheeze	All ages (asthmatics)	Linn et al. (1987, 1988, 1990) and Roger and al. (1985)	PM _{2.5}
Shortness of breath	Ages 7-12 (African American asthmatics)	Ostro et al. (1991)	PM _{2.5}
Work loss days	Ages 18-65	Ostro (1987)	PM ₁₀
Minor restricted activity days (MRAD)	Ages 18-65	Ostro and Rothschild (1989)	PM _{2.5} , O ₃
Restricted activity days	Ages 18-65	Ostro (1987)	PM _{2.5}

Table 3.9 Description of Hospital Admissions Studies for Respiratory Illnesses

<i>Study</i>	<i>Location and period</i>	<i>Sample Population</i>	<i>Endpoint</i>	<i>Pollutants</i>	<i>Main Findings</i>
Schwartz (1994a)	Detroit, MI, 1986-1989	Ages 64+	asthma (493), pneumonia (480-486), non-asthma COPD (491-492, 494-496)	O ₃ , PM ₁₀	PM 10 significant and O ₃ not significant for both pneumonia and COPD admissions in single-pollutant models.
Schwartz (1994b)	Minneapolis, MN, 1986-1989	Ages 64+	asthma (493), pneumonia (480-487), non-asthma COPD (491-492, 494-496)	O ₃ , PM ₁₀	No association found for asthma admissions. Both O ₃ and PM ₁₀ are significant for pneumonia and COPD admissions.
Schwartz (1994c)	Birmingham, AL, 1986, 1989	Ages 64+	pneumonia (480-487), COPD (490-496)	O ₃ , PM ₁₀	PM ₁₀ marginally significant for pneumonia admissions. O ₃ significant for COPD admissions. Both pollutants are significant for all respiratory admissions.
Burnett et al. (1997)	Toronto, Canada, 1992-1994	All ages	all respiratory (464-466, 480-486, 490-494, 496)	CO, O ₃ , PM _{2.5} , PM _{2.5-10} , PM ₁₀ , SO ₂	O ₃ linked to respiratory admissions, PM less strongly linked. CO, NO ₂ , and SO ₂ are not significant in two-pollutant models.

Table 3.9 Description of Hospital Admissions Studies for Respiratory Illnesses (continued)						
<i>Study</i>	<i>Location and period</i>	<i>Sample Population</i>	<i>Endpoint (ICD9 codes)</i>	<i>Pollutants</i>	<i>Main Findings</i>	
Moolgavkar et al. (1997)	Minneapolis, MN, Birmingham, AL, 1986-1991	Ages 64+	pneumonia (480-487), COPD (490-496), respiratory (480-487, 490-496)	CO, O ₃ , PM _{2.5} , PM _{2.5-10} , PM ₁₀ , SO ₂	O ₃ significant in multi-pollutant model for pneumonia admissions. No significant effect found for COPD admissions.	
Burnett et al. (1999)	Toronto, Canada, 1980-1994	All ages	asthma (493), respiratory infection (464,466,480-487,494), COPD, (490-492,496)	CO, O ₃ , PM _{2.5} , PM _{2.5-10} , PM ₁₀ , SO ₂	O ₃ , CO, PM _{2.5-10} significant for asthma admissions. O ₃ , NO ₂ , PM _{2.5} chosen in stepwise regression for respiratory infection. O ₃ and PM _{2.5-10} significant for COPD admissions.	
Sheppard et al. (1999)	Seattle, WA, 1987-1994	Ages 65+	Asthma (493)	CO, O ₃ , PM _{2.5} , PM _{2.5-10} , PM ₁₀ , SO ₂	In multi-pollutant models PM _{2.5} and CO are most significantly related to asthma admissions.	
Linn et al. (2000)	South Coast Air Basin, CA, 1992-1995	Ages 30+	asthma (493), COPD	CO, O ₃ , PM ₁₀	Pulmonary disease associated more with NO ₂ and PM ₁₀ than CO. High O ₃ concentrations seem to present less risk.	

Table 3.9 Description of Hospital Admissions Studies for Respiratory Illnesses (continued)					
Study	Location and period	Sample Population	Endpoint	Pollutants	Main Findings
Moolgavkar (2000)	Cook County, IL, Los Angeles County, CA, Maricopa County, AZ, 1987-1995	Ages 65+	COPD (490-496)	CO, O ₃ , PM _{2.5} , PM ₁₀ , SO ₂	In Cook and Maricopa counties there was a weak association between admissions and O ₃ , and no association with PM ₁₀ . In Los Angeles gaseous pollutants other than ozone were strongly associated with COPD admissions. PM was significant only in single-pollutant models.
Zanobetti et al. (2000)	Chicago, Cook County, IL, 1985-1994	All ages	COPD (490-496 except 493), pneumonia (480-487), asthma (493), acute bronchitis (466), acute respiratory illness (460-466)	PM ₁₀	Diagnosis of conduction disorders or dysrhythmias increased the associations between hospital admissions for COPD and pneumonia and PM ₁₀ . Persons with asthma had twice the risk of PM ₁₀ -related pneumonia admissions, and persons with heart failure had twice the risk of PM ₁₀ -induced COPD admissions
Lin et al. (2003)	Toronto, Canada, 1981-1993	Ages 6-12	asthma (493)	CO, O ₃ , PM _{2.5} , PM _{2.5-10} , PM ₁₀ , SO ₂	Significant acute effect of CO was found in boys, and SO ₂ showed significant effects of chronic exposure in girls. NO ₂ was positively associated with admissions in both sexes. O ₃ was not significant.
Chen et al. (2004)	Vancouver, Canada, 1995-1999	Ages 65+	COPD (490-492, 494, 496)	CO, O ₃ , PM _{2.5} , PM _{2.5-10} , PM ₁₀ , SO ₂	PM measures had a positive association with hospital admissions for COPD. The associations were no longer significant when NO ₂ was included in the models.

Table 3.10 Description of Hospital Admissions Studies for Cardiovascular Illnesses						
<i>Study</i>	<i>Location and period</i>	<i>Sample Population</i>	<i>Endpoint (ICD9 codes)</i>	<i>Pollutants</i>	<i>Main Findings</i>	
Schwartz and Morris (1995)	Detroit, MI, 1986-1989	Ages 64+	ischemic heart disease (410-414), dysrhythmias (427), congestive heart failure (428)	CO, O ₃ , PM ₁₀ , SO ₂	PM ₁₀ and Co significant in two-pollutant models for ischemic heart disease. No significant effect found for dysrhythmias. In two-pollutant models PM ₁₀ is significant for congestive heart failure admissions.	
Burnett et al. (1997)	Toronto, Canada, 1992-1994	All ages	cardiac (410-414, 427-428)	CO, O ₃ , PM _{2.5} , PM _{2.5-10} , PM ₁₀ , SO ₂	Significant affect found for O ₃ , and to a lesser extent, for PM. Other pollutants are not significant.	
Schwartz(1997)	Tucson AZ, 1988-1990	Ages 64+	cardiovascular disease (390-429)	CO, O ₃ , PM ₁₀ , SO ₂	CO and PM ₁₀ significant in two-pollutant models. No effect seen for other pollutants.	
Burnett et al. (1999)	Toronto, Canada, 1980-1994	All ages	ischemic heart disease (410-414), dysrhythmias (427), congestive heart failure (428)	CO, O ₃ , PM _{2.5} , PM _{2.5-10} , PM ₁₀ , SO ₂	NO ₂ and SO ₂ significant for ischemic heart disease. O ₃ , CO, and PM _{2.5} significant for dysrhythmias. NO ₂ and CO significant for congestive heart failure.	

Table 3.10 Description of Hospital Admissions Studies for Cardiovascular Illnesses (continued)					
Study	Location and period	Sample Population	Endpoint (ICD9 codes)	Pollutants	Main Findings
Schwartz (1999)	8 U.S. counties, 1988-1990	Ages 65+	cardiovascular disease (390-429)	CO, PM10	Both pollutants significant in two-pollutant models
Linn et al.(2000)	South Coast Air Basin, CA, 1992-1995	Ages 30+	congestive heart failure, myocardial infarction, cardiac arrhythmia	CO, O3, PM10	CO, NO2, and to a lesser extent PM10 showed consistently significant relationship to cardiovascular admissions. O3 was negatively associated or not significant.
Zanobetti et al. (2000)	Chicago, Cook County, IL, 1985-1994	All ages	cardiovascular disease (390-429), myocardial infarction (410), congestive heart failure (428), conduction disorders (426), dysrhythmias (427)	PM10	PM10-related hospital admissions for cardiovascular diseases were almost doubled in subjects with concurrent respiratory infections.
Mann et al. (2002)	South Coast Air Basin, CA, 1988-1995	All ages	ischemic heart disease (410-414)	CO, O3, PM10	CO was associated with same-day IHD admissions in persons with secondary diagnosis of arrhythmia.
Koken et al. (2003)	Denver, Colorado, 1993-1997	Ages 65+	Pulmonary Heart Disease, Coronary Atherosclerosis, Congestive Heart Failure, Cardiac Dysrhythmias	CO, O3, PM10, SO2	SO2 appears to be related to increased hospital stays for cardiac dysrhythmias, and CO is associated with congestive heart failure. No association was found with PM or NO2.

3.3 VALUATION OF HEALTH BENEFITS

3.3.1 Monetizing mortality benefits

Environmental economics developed a number of methods for estimating health benefits from avoided air pollution. The most popular primary methods are described below:

Contingent valuation method (CVM) is a survey-based method to determine willingness-to-pay (WTP) for a hypothetical change in environmental effects.

Averting behavior is a method to infer WTP from the costs and effectiveness of actions taken to avoid a negative environmental effect.

Cost-of-illness (COI) or damage costs methods involve estimating direct costs (such as, medical expenses) and indirect costs (for example, forgone earnings) of an environmental effect.

Hedonic methods estimate WTP for an environmental amenity by inferring its value from the market price of another (but in some sense related) good. For example, the *hedonic property values* method estimates a marginal WTP function based on an estimated relationship between housing prices and housing attributes (that include environmental amenities, such as good visibility or air quality). In contrast, the *hedonic wages* method estimates the value of environmental amenities from a worker's WTA a higher salary to compensate for exposure to higher levels of risk on the job.

Mortality benefits are most often monetized using a Value of Statistical Life (VSL) estimate. VSL is a measure of the WTP for reductions in the risk of premature death aggregated over the population experiencing the potential risk reduction. For example, if each person out of one million people is willing to pay five dollars for a 1:1,000,000 reduction in mortality risk, then on average one life is saved, and hence the value of VSL is \$5,000,000. (EPA relies on a composite VSL estimate based on 26 VSL studies – 21 of which use the hedonic wage method, and 5 use CVM).

Despite its widespread use, the VSL concept has been subject to criticism. One shortcoming of the VSL is highlighted when one considers the difference between mortality due to acute (short-term) and mortality due to chronic (long-term) exposure. One of the basic assumptions underlying the VSL approach is that that equal increments in fatality risk are valued equally irrespective of the initial risk. This assumption is defensible only if the prior risk is small. Some experts have suggested that it is not appropriate to estimate mortality that is the result of acute exposure, because people affected usually have a pre-existing disease and a relatively high prior risk of mortality.

Alternative measures to VSL include the Value of Statistical Life Years (VSLY) lost or saved. For example, if pollution abatement saves one person with average life expectancy of fifty more years, then we say that the policy results in fifty life years extended. VSLY

can be interpreted as an age-specific VSL. Another alternative to VSL is the Quality Adjusted Life Years (QALY) measure. QALY adjusts life-years extended for the quality of life during those years. To estimate QALY, we need information about the path and duration of health states, as well as we have to choose weights for the different health states.

The primary approach of estimating VSL has been the use of hedonic wage and hedonic price models that examine the equilibrium risk choices. The observed market decisions (measured by wages and prices) reflect the joint influence of supply and demand in the market. Most of the empirical literature has concentrated on valuing mortality risk by estimating compensating differentials for on-the-job risk exposure in labor markets. Viscusi and Aldy (2003) provide a meta-analysis of the extensive literature of VSL based on estimates using U.S. labor market data from the last three decades. The main results of these studies are summarized in Table 3.11. Estimates of VSL typically range from \$4 million to \$9 million.

The wage-risk relationship is typically estimated by regressing the observed wage of individuals on a vector of personal characteristics, a vector of job characteristics, fatality and non-fatality risk associated with the job, and the workers' compensation benefits payable for a job injury suffered by the worker. An important methodological question is the choice of a risk measure. An ideal risk measure would reflect both the employee's perception of on-the-job fatality risk and the employer's perception of such risk. Suitable measures of the subjective risk are rarely available, and therefore the standard approach in the literature has been the use of industry-specific or occupation-specific risk measures (e.g. average number or rate of fatalities over a period of several years).

Hedonic wage studies vary in several aspects, such as, their labor market coverage (entire labor force vs. specific occupations), geographic coverage (entire country vs. specific states or regions), class of workers (e.g. blue-collar workers only), gender, union-status of workers (e.g. union-members only), as well as, the measure of mortality risk used.

Table 3.11 Summary of Labor Market Studies of the Value of a Statistical Life, United States					
Author (Year)	Sample	Risk Variable	Mean Risk	Average Income Level (2000 US\$)	Implicit VSL (millions 2000 US\$)
Smith (1974)	Current Population Survey (CPS) 1967 Census of Manufactures 1963 U.S. Census 1960 Employment and Earnings 1963	Bureau of Labor Statistics (BLS) 1966 1967	0.000125	\$29,029	\$9.2
Thaler and Rosen (1975)	Survey of Economic Opportunity 1967	Society of Actuaries 1967	0.001	\$34,663	\$1.0
Smith (1976)	CPS 1967 1973	BLS 1966 1967 1970	0.0001	\$31,027	\$5.9
Viscusi (1978a 1979)	Survey of Working Conditions 1969-1970 (SWC)	BLS 1969 subjective risk of job (SWC)	0.0001	\$31,842	\$5.3
Brown (1980)	National Longitudinal Survey of Young Men 1966-71 1973	Society of Actuaries 1967	0.002	\$49,019	\$1.9
Viscusi (1981)	Panel Study of Income Dynamics (PSID) 1976	BLS 1973-1976	0.0001	\$22,618	\$8.3
Olson (1981)	CPS 1978	BLS 1973	0.0001	\$36,151	\$6.7
Arnould and Nichols (1983)	U.S. Census 1970	Society of Actuaries 1967	0.001	NA	\$0.5-\$1.3
Butler (1983)	S.C. Workers' Compensation Data 1940-69	S.C. Workers' Compensation Claims Data	0.00005	\$22,713	\$1.3

Table 3.11 Summary of Labor Market Studies of the Value of a Statistical Life, United States (continued)

<i>Author (Year)</i>	<i>Sample</i>	<i>Risk Variable</i>	<i>Mean Risk</i>	<i>Average Income Level (2000 US\$)</i>	<i>Implicit VSL (millions 2000 US\$)</i>
Low and McPheters (1983)	International City Management Association 1976 (police officer wages)	Constructed a risk measure from DOJ/FBI police officers killed data 1972-75 for 72 cities	0.0003	\$33,172	\$1.4
Dorsey and Walzer (1983)	CPS May 1978	BLS 1976	0.000052	\$21,636	\$11.8- \$12.3
Leigh and Folsom (1984)	PSID 1974 Quality of Employment Survey (QES) 1977	BLS	0.0001	\$29,038 \$36,946	\$10.1-\$13.3
Smith and Gilbert (1984 1985)	CPS 1978	BLS 1975	NA	NA	\$0.9
Dillingham and Smith (1984)	CPS May 1979 BLS industry data 1976 1979	NY Workers' Comp Data 1970	0.000082	\$29,707	\$4.1-\$8.3
Dillingham (1985)	QES 1977	BLS 1976 NY Workers' Compensation data 1970	0.000008 0.00014	\$26,731	\$1.2 \$3.2-\$6.8
Leigh (1987)	QES 1977 CPS 1977	BLS	NA	NA	\$13.3
Moore and Viscusi (1988a)	PSID 1982 BLS 1972-1982	NIOSH National Traumatic Occupational Fatality (NTOF) Survey 1980-1985	0.00005 0.00008	\$24,931	\$3.2 \$9.4

Source: Viscusi and Aldy (2003), Table 2, p.88

Half of the studies reviewed in Viscusi and Aldy (2003) provide estimates that range from \$5 million to \$12 million (in 2000 dollars). Estimates below \$5 million usually are reported by studies that use the Society of Actuaries data, which contains data on workers that self-selected themselves to jobs that are much riskier than average. On the other hand, studies that report estimates above \$12 million tend to estimate the wage-risk relationship indirectly. Viscusi and Aldy (2003) regard the median estimate of \$7 million from the above table as the most reliable.

These values of VSL using hedonic wage methods are similar to those generated by U.S. product market and housing market studies.

When transferring the estimates of VSL to non-labor market contexts, as is the case of cost-benefit analysis of the RPS, we must make sure that the preferences of the study population and the populations in the policy context are similar. Other factors that may

influence the transfer of VSL estimates are the age and income distribution of the study population. In addition to finding a positive association between income and VSL, Viscusi and Aldy (2003) also found a statistically significant relationship between union-status of workers and VSL.

3.3.2 Valuation of Morbidity Benefits

There are a number of health effects (endpoints) that can be quantified (U.S. EPA, 1989), but difficult to value in monetary terms. These effects include, for example, reduced lung function. Currently, there are no studies available on the economic valuation of changes in lung function. One reason is that there is no clear connection between lung function and the economic well-being of an individual. Reduced lung function is typically associated with other symptoms, such as cough or asthma attacks, and it is unclear whether a temporary decrement in lung function would go unnoticed without the related symptoms, and hence it may have no economic value.

Several endpoints reported in the literature overlap with each other, for example the various measures of restricted activity, or their definitions are not unique. Therefore, one must be careful not to include a combination of endpoints that could lead to double counting of benefits.

In a number of studies, it has been found that the different methods, mentioned in the previous sections, generate systematically different estimates for morbidity effects. The following table contains estimates of WTP and cost-of-illness for various health effects. For each effect, both estimates are from the same source. We can conclude from these results, that people's WTP for avoiding certain symptoms usually exceeds the cost of that symptom by an order of a magnitude. On the other hand there does not seem to be a systematic relationship between the two estimates.

Table 3.12 Comparison of the value of morbidity effects using different valuation methods				
Symptoms	WTP method	WTP \$1996	Individual COI \$1996	
Berger et al. (1987) Dollar value for one symptom day				
Cough	CVM	\$114.74	\$18.38	
Sinus Congestion	CVM	\$41.26	\$10.25	
Throat Congestion	CVM	\$66.34	\$21.55	
Itchy Eyes	CVM	\$73.21	\$21.99	
Heavy Drowsiness	CVM	\$214.44	\$2.72	
Headache	CVM	\$164.16	\$5.21	
Nausea	CVM	\$72.30	\$3.78	
All Symptoms	CVM	\$121.76	\$5.93	
Chestnut et al. (1988, 1996) Dollar value for one episode				
Symptoms	WTP method	WTP \$1996	Individual COI \$1996	
Angina Episodes	AB	\$54.40	\$18.54	
Angina Episodes	CVM	\$57.26	\$18.54	
Angina Episodes	CVM	\$60.13	\$18.54	
Angina Episodes	CVM	\$147.45	\$18.54	
Dickie-Gerking (1991) Dollar value for a reduction to zero days/year of ozone				
Health effects of ozone for concentration	WTP method	WTP \$1996	Medical expenses \$1996	
Over 12 pphm	AB	\$138.53	\$36.45	
Over 12 pphm	AB	\$167.69	\$84.57	
Over 12 pphm	AB	\$249.35	\$67.08	
Over 12 pphm	AB	\$304.76	\$160.40	
Over 9 pphm	AB	\$249.35	\$59.79	
Over 9 pphm	AB	\$298.93	\$131.24	
Over 9 pphm	AB	\$380.58	\$94.78	
Over 9 pphm	AB	\$457.87	\$215.81	
Rowe-Chestnut (1985)				
Asthma Severity	WTP method	WTP \$1996	\$1996 Medical expenses	Foregone Earnings \$1996
Asthma Severity	CVM	\$ 631.70	\$196.91	\$116.57
Asthma Severity	CVM	\$ 919.98	\$196.91	\$116.57
Asthma Severity	CVM	\$ 697.86	\$70.89	\$116.57
Source: U.S. EPA (2000)				

Valuation of hospital admissions avoided

The typical approach for valuing the avoided incidence of hospital admissions is through the use of COI method (U.S. EPA, 1999). Well-developed and detailed estimates of hospitalization of the health effects are readily available (U.S. EPA, 2004). COI estimates should be obtained for each health effects for which dose-response functions are available. Valuation estimates typically have two components: cost of hospital stay, and lost earnings due to hospitalization. As mentioned above, COI method underestimates WTP by a factor of at least three. However, there are currently no studies available that would estimate WTP directly.

CONCENTRATION-RESPONSE FUNCTIONS FOR OZONE (O₃)

Table 3.13 CR-Functions for Ozone (O₃) Health Endpoint: Onset of asthma					
CR-function: $\Delta \text{chronic asthma} = - \left[\frac{y_1}{(1 - y_1) \cdot e^{\beta \Delta O_3} + y_1} - y_1 \right] \cdot \text{pop}$ y_1 = occurrence rate of chronic asthma β = O ₃ coefficient ΔO_3 = change in O ₃ concentrations (ppm) pop = population sample					
Summary of Estimates					
Study	Location	Sample Population	Other Pollutants	Ozone Concentration Measure	β^1
McDonnell et al. (1999)	California	Non-asthmatic males ages 27+	None	Annual average 8-hour O ₃	0.0277 (0.0135)
McConnell et al. (2002)	Southern California	Children ages 9-16	PM ₁₀ , NO ₂	Annual 10:00h to 18:00h mean ozone concentration	0.0904 (0.0213)
Notes: 1 Standard error of β in brackets.					

Table 3.14 CR-Functions for Ozone (O₃) Health Endpoint: Hospital admissions – pneumonia					
CR-function: $\Delta \text{pneumonia admissions} = - \left[y_1 \cdot (e^{\beta \cdot \Delta O_3} - 1) \right] \cdot \text{pop}$ y_1 = daily admissions rate for pneumonia per person β = O ₃ coefficient ΔO_3 = change in O ₃ concentrations (ppm) pop = population sample					
Summary of Estimates					
Study	Location	Sample Population	Other Pollutants	Ozone Concentration Measure	β^1
Moolgavkar et al. (1997)	Minneapolis, MN	Population ages 65+	SO ₂ , NO ₂ , PM ₁₀	Daily average O ₃ concentration	0.00370 (0.00103)
Schwartz (1994b)	Detroit, MI	Population ages 65+	PM ₁₀	Daily average O ₃ concentration	0.00521 (0.0013)
Schwartz (1994c)	Minneapolis, MN	Population ages 65+	PM ₁₀	Daily average O ₃ concentration	0.00280 (0.00071)
Notes: 1. Standard error of β in brackets.					

Table 3.15 CR-Functions for Ozone (O₃)					
Health Endpoint: Mortality					
CR-function: $\Delta\text{mortality} = - \left[y_1 \cdot (e^{\beta \cdot \Delta O_3} - 1) \right] \cdot \text{pop}$ y_1 = non-accidental deaths per person of any age β = O ₃ coefficient ΔO_3 = change in O ₃ concentrations (ppm) pop = population sample					
Summary of Estimates					
Study	Location	Sample Population	Other Pollutants	Ozone Concentration Measure	β^1
Kinney et al. (1995)	Los Angeles, CA	Population of all ages	PM ₁₀	Daily one-hour maximum O ₃	0.00010 (0.000214)
Moolgavkar et al. (1995)	Philadelphia, PA	Population of all ages	SO ₂ , TSP	Daily average ozone	0.000611 (0.000216)
Ito and Thurston (1996)	Chicago, IL	Population of all ages	PM ₁₀	Daily one-hour maximum O ₃	0.00068 (0.00029)
Kelsall et al. (1997)	Philadelphia, PA	Population of all ages	CO, NO ₂ , SO ₂ , TSP	Daily average ozone	0.000936 (0.000312)
Golberg et al. (2001)	Montréal, Québec	Population of all ages	CO, NO ₂ , NO, SO ₂ , PM _{2.5}	Daily average ozone	0.002056 (0.000475)
Notes: 1. Standard error of β in brackets.					

Table 3.16 CR-Functions for Ozone (O₃)					
Health Endpoint: Emergency room visits for asthma					
CR-function: $\Delta\text{asthma-related ER visits} = \frac{\beta}{\text{BasePop}} \Delta O_3 \cdot \text{pop}$ BasePop = baseline population β = O ₃ coefficient ΔO_3 = change in O ₃ concentrations (ppm) pop = population sample					
Summary of Estimates					
Study	Location	Sample Population	Other Pollutants	Ozone Concentration Measure	β^1
Cody et al. (1992)	Northern New Jersey	Population of all ages	None	Daily five-hour average O ₃	0.0203 (0.00717)
Weisel et al. (1992)	Northern New Jersey	Population of all ages	None	Daily five-hour average O ₃	0.0443 (0.00723)
Stieb et al. (1996)	New Brunswick, Canada	Population of all ages	None	Daily one-hour maximum O ₃	0.0035 (0.0018)
Notes: 1. Standard error of β in brackets.					

Table 3.17 CR-Functions for Ozone (O₃) Health Endpoint: Hospital admissions - all respiratory causes					
CR-function: $\Delta \text{all respir. admissions} = - \left[y_1 \cdot (e^{\beta \cdot \Delta O_3} - 1) \right] \cdot \text{pop}$ y_1 = daily admissions rate for all respiratory causes per person β = O ₃ coefficient ΔO_3 = change in O ₃ concentrations (ppm) pop = population sample					
Summary of Estimates					
Study	Location	Sample Population	Other Pollutants	Ozone Concentration Measure	β^1
Thurston et al. (1994)	Toronto, Canada	Population of all ages	PM _{2.5}	Daily one-hour maximum O ₃ concentrations	0.00250 (9.71E-9)
Schwartz (1995)	New Haven, CT	Population ages 65+	PM ₁₀	Daily average O ₃	0.00265 (0.00140)
Schwartz (1995)	Tacoma, WA	Population ages 65+	PM ₁₀	Daily average O ₃ concentrations	0.00715 (0.00257)
Burnett et al. (1997)	Toronto, Canada	Population of all ages	PM _{2.5} , PM ₁₀ , NO ₂ , SO ₂	Daily one-hour maximum O ₃ concentrations	0.00498 (0.00106)
Notes: 1. Standard error of β in brackets.					

Table 3.18 CR-Functions for Ozone (O₃) Health Endpoint: Hospital admissions for chronic obstructive pulmonary disease (COPD)					
CR-function: $\Delta \text{COPD admissions} = - \left[y_1 \cdot (e^{\beta \cdot \Delta O_3} - 1) \right] \cdot \text{pop}$ y_1 = daily admissions rate for COPD per person β = O ₃ coefficient ΔO_3 = change in O ₃ concentrations (ppm) pop = population sample					
Summary of Estimates					
Study	Location	Sample Population	Other Pollutants	Ozone Concentration Measure	β^1
Schwartz (1994b)	Detroit, MI	Population 65+	PM ₁₀	Daily average O ₃ concentration	0.00549 (0.00205)
Moolgavkar et al. (1997)	Minneapolis, MN	Population 65+	CO, NO ₂	Daily average O ₃ concentration	0.00274 (0.00170)
Burnett et al. (1999)	Toronto, Canada	Population of all ages	CO, PM _{2.5} , PM ₁₀	Daily average O ₃ concentration	0.00303 (0.00110)
Notes: 1. Standard error of β in brackets.					

Table 3.19 CR-Functions for Ozone (O₃) Health Endpoint: Hospital admissions for asthma					
CR-function: $\Delta \text{asthma admissions} = - \left[y_1 \cdot (e^{\beta \cdot \Delta O_3} - 1) \right] \cdot \text{pop}$ y_1 = daily asthma admission rate per person β = O ₃ coefficient ΔO_3 = change in O ₃ concentrations (ppb) pop = sample population					
Summary of Estimates					
Study	Location	Sample Population	Other Pollutants	Ozone Concentration Measure	β^1
Burnett et al. (1999)	Toronto, Canada	Population of all ages	CO, PM _{2.5} , PM ₁₀	Daily average O ₃ concentration	0.00250 (0.000718)
Notes: 1. Standard error of β in brackets.					

Table 3.20 CR-Functions for Ozone (O₃) Health Endpoint: Hospital admissions for respiratory infection					
CR-function: $\Delta \text{respiratory infections admissions} = - \left[y_1 \cdot (e^{\beta \cdot \Delta O_3} - 1) \right] \cdot \text{pop}$ y_1 = daily respiratory infections admission rate per person β = O ₃ coefficient ΔO_3 = change in O ₃ concentrations (ppb) pop = sample population					
Summary of Estimates					
Study	Location	Sample Population	Other Pollutants	Ozone Concentration Measure	β^1
Burnett et al. (1999)	Toronto, Canada	Population of all ages	PM _{2.5} , NO ₂	Daily average O ₃ concentration	0.00198 (0.000520)
Notes: 1. Standard error of β in brackets.					

Table 3.21 CR-Functions for Ozone (O₃) Health Endpoint: Hospital admissions for cardiac					
CR-function: $\Delta \text{cardiac admissions} = - \left[y_1 \cdot (e^{\beta \cdot \Delta O_3} - 1) \right] \cdot \text{pop}$ y_1 = daily cardiac admission rate per person β = O ₃ coefficient ΔO_3 = change in O ₃ concentrations (ppb) pop = sample population					
Summary of Estimates					
Study	Location	Sample Population	Other Pollutants	Ozone Concentration Measure	β^1
Burnett et al. (1997)	Toronto, Canada	Population of all ages	PM _{2.5} , PM ₁₀	Daily 12-hour average	0.00531 (0.00142)
Notes: 1. Standard error of β in brackets.					

Table 3.22 CR-Functions for Ozone (O₃) Health Endpoint: Hospital admissions for dysrhythmias					
CR-function: $\Delta \text{cardiac admissions} = - \left[y_1 \cdot (e^{\beta \cdot \Delta O_3} - 1) \right] \cdot \text{pop}$ y_1 = daily admission rate for dysrhythmias per person β = O ₃ coefficient ΔO_3 = change in O ₃ concentrations (ppb) pop = sample population					
Summary of Estimates					
Study	Location	Sample Population	Other Pollutants	Ozone Concentration Measure	β^1
Burnett et al. (1999)	Toronto, Canada	Population of all ages	CO, PM _{2.5}	Daily average O ₃	0.00168 (0.00103)
Notes: 1. Standard error of β in brackets.					

Table 3.23 CR-Functions for Ozone (O₃) Health Endpoint: Presence of any of 19 acute respiratory symptoms (ARD)					
CR-function: $\Delta \text{ARD2} \cong \beta_{PM10}^* \cdot \Delta O_3 \cdot \text{pop}$ β^* = first derivative of the stationary probability ΔO_3 = change in daily one-hour maximum O ₃ concentrations (ppb) pop = sample population					
Summary of Estimates					

Study	Location	Sample Population	Other Pollutants	Ozone Concentration Measure	β^1
Krupnick et al. (1990)	Glendora-Covina-Azusa, CA	Population aged 18-65	SO ₂ , COH	1-hour maximum	0.000137 (0.0000697)
Notes: 1. Standard error of β in brackets.					

Table 3.24 CR-Functions for Ozone (O ₃) Health Endpoint: Other minor health effects		
Self-reported asthma attacks	$\Delta \text{asthma attacks} = \left[\frac{y_1}{(1-y_1) \cdot e^{\Delta O_3 \beta} + y_1} - y_1 \right] \cdot \text{pop}$	
	y_1	= daily incidence of asthma attacks = 0.027
	β	= O ₃ coefficient = 0.00187
	ΔO_3	= change in daily one-hour maximum O ₃ concentration (ppb)
	pop	= population of asthmatics of all ages = 5.61% of the population of all ages
Respiratory and non-respiratory conditions resulting in minor restricted activity days (MRAD)	σ_β	= standard error of β = 0.000714
	$\Delta \text{MRAD} = - \left[y_1 \cdot (e^{\beta \cdot \Delta O_3} - 1) \right] \cdot \text{pop}$	
	y_1	= daily MRAD incidence per person = 0.02137
	β	= inverse-variance weighted PM _{2.5} coefficient = 0.00220
	ΔO_3	= change 2-week average of the daily 1-hour maximum O ₃ concentration (ppb)
	pop	= population 18-65
	σ_β	= standard error of β = 0.000658
Source: Costs and Benefits of the Clean Air Act, 1990-2010, Appendix D, Human Health Effects of Criteria Pollutants, EPA		

CONCENTRATION-RESPONSE FUNCTIONS FOR PARTICULATE MATTER (PM)

Table 3.25 CR-Functions for Particulate Matter (PM)

Health Endpoint: Mortality

$$\Delta \text{nonaccidental mortality} = - \left[y_1 \cdot (e^{-\beta \cdot \Delta PM_{2.5}} - 1) \right] \cdot \text{pop}$$

CR-function:

y_1 = non-accidental deaths per person
 β = $PM_{2.5}$ coefficient
 $\Delta PM_{2.5}$ = change in $PM_{2.5}$ concentrations (ppm)
 pop = population sample

Summary of Estimates

Study	Location	Sample Population	Other Pollutants	β^1
Dockery et al. (1993)	6 U.S. cities	Population ages 25+	None	0.0124 (0.00423)
Pope et al. (1995)	50 U.S. cities	Population ages 30+	None	0.006408 (0.001509)
HEI (2000)	6 U.S. cities	Population ages 25+	Air toxics	0.01327 (0.004408)
Pope et al. (2002)	50 U.S. cities	Population ages 30+	Sulfate, SO ₂ , NO ₂ , CO, O ₃	0.00583 (0.001963)

Notes: 1. Standard error of β in brackets.

Table 3.26 CR-Functions for Particulate Matter (PM)

Health Endpoint: Hospital Admissions – obstructive lung disease

$$\Delta \text{COPD hospital admissions} = - \left[y_1 \cdot (e^{-\beta \cdot \Delta PM} - 1) \right] \cdot \text{pop}$$

CR-function:

y_1 = hospital admissions for obstructive lung disease per person
 β = PM coefficient
 ΔPM = change in PM concentrations (ppm)
 pop = population sample

Summary of Estimates

Study	Location	Sample Population	PM Metric/Other Pollutants	β^1
Burnett et al. (1999)	Toronto, Canada	Population of all ages	$PM_{2.5-10}/O_3$	0.0019 (0.0003)

Table 3.27 CR-Functions for Particulate Matter (PM)
Health Endpoint: Hospital Admissions – chronic obstructive pulmonary disease (COPD)

$$\Delta \text{ COPD hospital admissions} = - \left[y_1 \cdot (e^{-\beta \cdot \Delta PM_{2.5}} - 1) \right] \cdot \text{pop}$$

CR-function:

y_1 = hospital admissions for COPD per person

β = PM coefficient

ΔPM = change in PM concentrations (ppm)

pop = population sample

Summary of Estimates

Study	Location	Sample Population	PM Metric/Other Pollutants	β^1
Schwartz (1994a)	Birmingham, AL	Population ages 65+	PM ₁₀ /None	0.00239 (0.000536)
Schwartz (1994b)	Detroit, MI	Population ages 65+	PM ₁₀ /O ₃	0.00202 (0.00059)
Schwartz (1994c)	Minneapolis, MN	Population ages 65+	PM ₁₀ /None	0.00451 (0.00138)
Moolgavkar et al. (1997)	Minneapolis, MN	Population ages 65+	PM ₁₀ /CO, O ₃	0.00088 (0.000777)
Moolgavkar (2000)	Los Angeles County	Population ages 65+	PM ₁₀ /None	0.00150 (0.000384)
Zanobetti et al. (2000)	Cook County, IL	Population ages 65+	PM ₁₀	0.007603 (0.003069)
Chen et al. (2004)	Vancouver, Canada	Population ages 65+	PM ₁₀ /None	0.01525 (0.004628)

Notes: 1. Standard error of β in brackets.

Table 3.28 CR-Functions for Particulate Matter (PM)
Health Endpoint: Hospital Admissions – pneumonia

$\Delta \text{ pneumonia hospital admissions} = - \left[y_1 \cdot (e^{-\beta \cdot \Delta PM_{2.5}} - 1) \right] \cdot \text{pop}$				
CR-function: y_1 = hospital admissions for pneumonia per person β = PM coefficient ΔPM = change in PM concentrations (ppm) pop = population sample				
Summary of Estimates				
Study	Location	Sample Population	PM Metric/Other Pollutants	β^1
Schwartz (1994a)	Birmingham, AL	Population ages 65+	PM ₁₀ /None	0.00174 (0.000838)
Schwartz (1994b)	Detroit, MI	Population ages 65+	PM ₁₀ /O ₃	0.00115 (0.00039)
Schwartz (1994c)	Minneapolis, MN	Population ages 65+	PM ₁₀ /O ₃	0.00157 (0.000677)
Moolgavkar et al. (1997)	Minneapolis, MN	Population ages 65+	PM ₁₀ /O ₃ , NO ₂ , SO ₂	0.000498 (0.000505)
Zanobetti et al. (2000)	10 U.S. cities	Population ages 65+	PM ₁₀ /O ₃ , SO ₂ , CO	0.00193 (0.00039)
Notes: 1. Standard error of β in brackets.				

CONCENTRATION-RESPONSE FUNCTIONS FOR CARBON MONOXIDE (CO)

Table 3.29 CR-Functions for Carbon Monoxide (CO)
Health Endpoint: Mortality

CR-function: $\Delta \text{mortality} = - \left[y_1 \cdot (e^{\beta \cdot \Delta CO} - 1) \right]$				
y_1 = Non-accidental deaths per person of any age β = CO coefficient ΔCO = change in CO concentrations (ppm) pop = population sample				
Summary of Estimates				
Study	Location	Sample Population	Other Pollutants	β^1
Saldiva et al. (1995)	São Paulo, Brazil	Ages 65+	None	0.015918 (0.00507)
Touloumi et al. (1996)	Athens, Greece	All ages	SO ₂	0.005511 (0.00167)
Burnett et al. (1998)	Toronto, Canada	All ages	TSP	0.0266 (0.0038)
Hong et al. (2002)	Seoul, South Korea	All ages	TSP	0.0890 (0.0256)

Table 3.30 CR-Functions for Carbon Monoxide (CO)				
Health Endpoint: Hospital admissions for asthma (ICD9-493)				
CR-function: $\Delta\text{asthma admissions} = -\left[y_1 \cdot (e^{\beta \cdot \Delta\text{CO}} - 1)\right] \cdot \text{pop}$				
y_1	=	daily hospital admissions rate for asthma per person		
β	=	CO coefficient		
ΔCO	=	change in average daily CO concentrations (ppm)		
pop	=	population sample		
Summary of Estimates				
Study	Location	Sample Population	Other Pollutants	β^1
Burnett et al. (1999)	Toronto, Canada	All ages	PM _{2.5} , PM ₁₀ , O ₃	0.0332 (0.00861)
Sheppard (1999)	Seattle, WA	Under age 65	PM _{2.5}	0.0528 (0.0185)
Linn et al. (2000)	Los Angeles, CA	Population ages 30+	PM ₁₀ , O ₃ , NO ₂	0.0280 (0.0100)
Lin et al. (2003)	Toronto, Canada	Boys ages 6-12	NO ₂ , SO ₂ , O ₃	0.1353 (0.0589)
Notes: 1.Standard error of β in brackets. 2.ICD = International Classification of Diseases				

Table 3.31 CR-Functions for Carbon Monoxide (CO)				
Health Endpoint: Hospital admissions for congestive heart failure				
CR-function: $\Delta\text{cong. heart failure admissions} = -\left[y_1 \cdot (e^{\beta \cdot \Delta\text{CO}} - 1)\right] \cdot \text{pop}$				
y_1	=	daily hospital admissions rate for congestive heart failure per person		
β	=	CO coefficient		
ΔCO	=	change in average daily CO concentrations (ppm)		
pop	=	population sample		
Summary of Estimates				
Study	Location	Sample Population	Other Pollutants	β^1
Schwartz and Morris (1995)	Detroit, MI	Ages 65+	PM ₁₀	0.0170 (0.00468)
Burnett et al. (1999)	Toronto, Canada	All ages	NO ₂	0.0340 (0.0163)
Koken et al. (2003)	Denver, CO	Ages 65+	NO ₂ , O ₃ , PM ₁₀	0.3328 (0.1681)
Notes: 1.Standard error of β in brackets. 2. ICD = International Classification of Diseases				

Table 3.32 CR-Functions for Carbon Monoxide (CO)				
Health Endpoint: Hospital admissions for cardiovascular disease				
CR-function: Δ cardiovascular admissions = $- \left[y_1 \cdot (e^{\beta \cdot \Delta CO} - 1) \right] \cdot \text{pop}$				
y_1	=	daily hospital admissions rate for cardiovascular disease per person		
β	=	CO coefficient		
Δ CO	=	change in average daily CO concentrations (ppm)		
pop	=	population sample		
Summary of Estimates				
Study	Location	Sample Population	Other Pollutants	β^1
Schwartz (1997)	Tucson, AZ	Ages 65+	PM ₁₀	0.0139 (0.00715)
Schwartz (1999)	8 U.S. counties	Ages 65+	PM ₁₀	0.0127 (0.00255)
Notes: 1. Standard error of β in brackets. 2. ICD = International Classification of Diseases				

Table 3.33 CR-Functions for Carbon Monoxide (CO)				
Health Endpoint: Hospital admissions for obstructive lung disease				
CR-function: Δobst. lung disease admissions = $- \left[y_1 \cdot (e^{\beta \cdot \Delta CO} - 1) \right] \cdot \text{pop}$				
y ₁	=	daily hospital admissions rate for obstructive lung disease per person		
β	=	CO coefficient		
ΔCO	=	change in average daily CO concentrations (ppm)		
pop	=	population sample		
Summary of Estimates				
Study	Location	Sample Population	Other Pollutants	β ¹
Moolgavkar ² (1997)	Minneapolis-St. Paul, MN	Population ages 65+	PM ₁₀ , O ₃	0.0573 (0.0329)
Burnett et al. (1999)	Toronto, Canada	Population of all ages	PM _{2.5} , PM ₁₀ , O ₃	0.0250 (0.0165)
Notes: 1. Standard error of β in brackets. 2. Health endpoint: COPD				

Table 3.34 CR-Functions for Carbon Monoxide (CO)				
Health Endpoint: Hospital admissions for dysrhythmias				
CR-function: $\Delta \text{dysrhythmias admissions} = - \left[y_1 \cdot (e^{\beta \cdot \Delta \text{CO}} - 1) \right] \cdot \text{pop}$				
y_1	=	daily hospital admissions rate for dysrhythmias per person		
β	=	CO coefficient		
ΔCO	=	change in average daily CO concentrations (ppm)		
pop	=	population sample		
Summary of Estimates				
Study	Location	Sample Population	Other Pollutants	β^1
Burnett et al. (1999)	Toronto, Canada	Population of all	PM _{2.5} , O ₃	0.0573

		ages		(0.0229)
Notes: 1.Standard error of β in brackets. 2. Health endpoint: COPD				

Table 3.35 CR-Functions for Carbon Monoxide (CO)				
Health Endpoint: Hospital admissions for obstructive lung disease				
CR-function: Δ ischemic heart disease admissions = $- \left[y_1 \cdot (e^{\beta \cdot \Delta CO} - 1) \right] \cdot \text{pop}$				
y_1	=	daily hospital admissions rate for dysrhythmias per person		
β	=	CO coefficient		
Δ CO	=	change in average daily CO concentrations (ppm)		
pop	=	population sample		
Summary of Estimates				
Study	Location	Sample Population	Other Pollutants	β^1
Schwartz and Morris (1995)	Detroit, MI	Population ages 65+	PM ₁₀	0.000467 (0.000435)
Notes: 1.Standard error of β in brackets. 2. Health endpoint: COPD				

CONCENTRATION-RESPONSE FUNCTIONS FOR NITROGEN DIOXIDE (NO₂)

Table 3.36 CR-Functions for Nitrogen Dioxide (NO ₂)				
Health Endpoint: Hospital admissions for respiratory conditions				
CR-function: Δall respiratory admissions = $- \left[y_1 \cdot (e^{\beta \cdot \Delta NO_2} - 1) \right] \cdot \text{pop}$				
y ₁	=	daily hospital admissions rate for respiratory conditions per person		
β	=	NO ₂ coefficient		
ΔNO ₂	=	change in average daily NO ₂ concentrations (ppm)		
pop	=	population sample		
Summary of Estimates				
Study	Location	Sample Population	Other Pollutants	β ¹
Burnett et al. (1997) ²	Toronto, Canada	Population of all ages	PM _{2.5} , PM ₁₀ , O ₃	0.00378 (0.00221)
Burnett et al. (1999) ³	Toronto, Canada	Population of all ages	PM _{2.5} , O ₃	0.00172 (0.000521)
Notes: 1.Standard error of β in brackets. 2. Health endpoint: all respiratory, NO ₂ measure: 12-hour average 3.Health endpoint: respiratory infection				

Table 3.37 CR-Functions for Nitrogen Dioxide (NO₂)**Health Endpoint: Hospital admissions for pneumonia**

CR-function: $\Delta \text{pneumonia admissions} = - \left[y_1 \cdot (e^{\beta \cdot \Delta \text{NO}_2} - 1) \right] \cdot \text{pop}$

y_1 = daily hospital admissions rate for pneumonia per person
 β = NO₂ coefficient
 ΔNO_2 = change in average daily NO₂ concentrations (ppm)
 pop = population sample

Summary of Estimates

Study	Location	Sample Population	Other Pollutants	β^1
Moolgavkar (1997)	Minneapolis-St. Paul, MN	Population ages 65+	PM ₁₀ , O ₃ , SO ₂	0.00172 (0.00125)

Notes: 1.Standard error of β in brackets.

Table 3.38 CR-Functions for Nitrogen Dioxide (NO₂)**Health Endpoint: Hospital admissions for congestive heart failure**

CR-function: $\Delta \text{cong. heart failure admissions} = - \left[y_1 \cdot (e^{\beta \cdot \Delta \text{NO}_2} - 1) \right] \cdot \text{pop}$

y_1 = daily hospital admissions rate for congestive heart failure per person
 β = NO₂ coefficient
 ΔNO_2 = change in average daily NO₂ concentrations (ppm)
 pop = population sample

Summary of Estimates

Study	Location	Sample Population	Other Pollutants	β^1
Burnett et al. (1999)	Toronto, Canada	Population of ages	PM _{2.5} , O ₃	0.00264 (0.000769)

Notes: 1.Standard error of β in brackets.

Table 3.39 CR-Functions for Nitrogen Dioxide (NO₂)**Health Endpoint: Hospital admissions for ischemic heart disease**

CR-function: $\Delta \text{ischemic heart disease admissions} = - \left[y_1 \cdot (e^{\beta \cdot \Delta \text{NO}_2} - 1) \right] \cdot \text{pop}$

y_1 = daily hospital admissions rate for ischemic heart disease per person
 β = NO₂ coefficient
 ΔNO_2 = change in average daily NO₂ concentrations (ppm)
 pop = population sample

Summary of Estimates

Study	Location	Sample Population	Other Pollutants	β^1
Burnett et al. (1999)	Toronto, Canada	Population of ages	SO ₂	0.00318 (0.000521)

Notes: 1.Standard error of β in brackets.

CONCENTRATION-RESPONSE FUNCTIONS FOR SULFUR DIOXIDE (SO₂)

Table 3.40 CR-Functions for Sulfur Dioxide (SO₂)
Health Endpoint: Mortality

CR-function: $\Delta \text{mortality} = -[y_1 \cdot (e^{\beta \cdot \Delta \text{SO}_2} - 1)]$

y_1 = Non-accidental deaths per person of any age
 β = SO₂ coefficient
 ΔSO_2 = change in SO₂ concentrations (ppb)
 pop = population sample

Summary of Estimates

Study	Location	Sample Population	Other Pollutants	β^1
Saldiva et al. (1995)	São Paulo, Brazil	Ages 65+	None	0.005204 (0.002229)
Touloumi et al. (1996)	Athens, Greece	All ages	CO	0.00404 (0.00006)
Hong et al. (2002)	Seoul, South Korea	All ages	TSP	0.003343 (0.001126)

Table 3.41 CR-Functions for Sulfur Dioxide (SO₂)
Health Endpoint: Emergency department (ED) admissions for cardiac

CR-function: $\Delta \text{cardiac ED admissions} = -[y_1 \cdot (e^{\beta \cdot \Delta \text{SO}_2} - 1)]$

y_1 = cardiac ED admissions per person
 β = SO₂ coefficient
 ΔSO_2 = change in SO₂ concentrations (ppb)
 pop = population sample

Summary of Estimates

Study	Location	Sample Population	Other Pollutants	β^1
Stieb et al. (2000)	Saint John, Canada	All ages	O ₃ , PM ₁₀ , PM _{2.5} , SO ₄	0.00201 (0.000664)

Table 3.42 CR-Functions for Sulfur Dioxide (SO₂)
Health Endpoint: Emergency department (ED) admissions for respiratory causes

CR-function: $\Delta \text{cardiac ED admissions} = -[y_1 \cdot (e^{\beta \cdot \Delta \text{SO}_2} - 1)]$

y_1 = respiratory ED admissions per person
 β = SO₂ coefficient
 ΔSO_2 = change in SO₂ concentrations (ppb)
 pop = population sample

Summary of Estimates

Study	Location	Sample Population	Other Pollutants	β^1
Stieb et al. (2000)	Saint John, Canada	All ages	O ₃ , PM ₁₀ , PM _{2.5} , SO ₄	0.001527 (0.000460)

Table 3.43 CR-Functions for Sulfur Dioxide (SO₂) Health Endpoint: Hospital admissions for asthma				
CR-function: $\Delta \text{Asthma hospital admissions} = -[y_1 \cdot (e^{\beta \cdot \Delta SO_2} - 1)]$ y_1 = hospital admissions for asthma per person β = SO ₂ coefficient ΔSO_2 = change in SO ₂ concentrations (ppb) pop = population sample				
Summary of Estimates				
Study	Location	Sample Population	Other Pollutants	β^1
Lin et al. (2003)	Toronto, Canada	Children ages 6-12	PM _{2.5}	0.035266 (0.012383)

Table 3.44 CR-Functions for Sulfur Dioxide (SO₂) Health Endpoint: Hospital admissions for COPD				
CR-function: $\Delta \text{COPD hospital admissions} = -[y_1 \cdot (e^{\beta \cdot \Delta SO_2} - 1)]$ y_1 = hospital admissions for COPD per person β = SO ₂ coefficient ΔSO_2 = change in SO ₂ concentrations (ppb) pop = population sample				
Summary of Estimates				
Study	Location	Sample Population	Other Pollutants	β^1
Moolgavkar (2000)	Los Angeles County	Ages 0-19	None	0.0154 (0.0011)
Moolgavkar (2000)	Los Angeles County	Ages 20-64	None	0.0125 (0.0008)
Moolgavkar (2000)	Los Angeles County	Ages 65+	None	0.0113 (0.0010)
Moolgavkar (2000)	Maricopa County	Ages 65+	None	0.0138 (0.0019)

Table 3.45 CR-Functions for Sulfur Dioxide (SO₂) Health Endpoint: Hospital admissions for ischemic heart disease				
CR-function: $\Delta \text{ischemic heart disease hospital admissions} = -[y_1 \cdot (e^{\beta \cdot \Delta SO_2} - 1)]$ y_1 = hospital admissions for ischemic heart disease per person β = SO ₂ coefficient ΔSO_2 = change in SO ₂ concentrations (ppb) pop = population sample				
Summary of Estimates				
Study	Location	Sample Population	Other Pollutants	β^1
Burnett et al. (1999)	Toronto, Canada	All ages	None	0.0009 (0.0001)

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4. INDIRECT BENEFITS TO HUMANS THROUGH ECOSYSTEMS

Air pollution can cause significant ecological damages. Table 4.1 below identifies the most important direct and indirect effects of air pollution.

Table 4.1
Ecological effects of air pollution

Pollutant Class	Major Pollutants and Precursors	Short-term effects	Long-term effects
Acidic Deposition	Sulfuric acid, nitric acid Precursors: SO ₂ , NO _x	Direct toxic effect to plant leaves and aquatic organisms	Progressive deterioration of soil quality and acidification of surface waters
Nitrogen Deposition	NO _x		Saturation of terrestrial ecosystems with nitrogen. Progressive nitrogen enrichment of coastal estuaries.
Hazardous Air Pollutants (HAPs)	Mercury and dioxins	Direct toxic effects to animals.	Accumulation of mercury and dioxin in the food chain.
Ozone	Tropospheric ozone	Direct toxic effects to plant leaves.	Alteration of ecosystem-wide energy flow and nutrient cycling.
Source: U.S. EPA (1999)			

The first step in valuing ecosystem benefits of reduced air pollution is to identify measurable endpoints. Freeman (1997) identified the following categories of ecosystem services to humankind:

1. Material inputs into economic activity (fossil fuels, minerals, animals)
2. Life-support services (breathable air, livable climate)
3. Environmental amenities used for recreation
4. Processing of waste products discharged into the environment

Available valuation methods can measure only some of these benefits: material inputs and the value of environmental amenities used for active recreation. The most important ecological effects with identifiable service flow impacts are summarized in Table 4.2.

Table 4.2 Ecological effects with identifiable and measurable human service flows		
Pollutant Class	Ecosystem effect	Service flow impacted
Acidification (H ₂ SO ₄ , HNO ₃)	High-elevation forest acidification resulting in dieback Freshwater acidification resulting in fresh water organism (e.g. fish) population decline. Changes in biodiversity in terrestrial and aquatic ecosystems	Forest esthetics Recreational Fishing Existence/Non-use values of biodiversity
Nitrogen Saturation and Eutrophication (NO _x)	Freshwater acidification resulting in fresh water organism (e.g. fish) population decline. Estuarine eutrophication causing oxygen depletion and changes in nutrient cycling Changes in biodiversity in terrestrial and aquatic ecosystems	Recreational Fishing Recreational and commercial fishing Existence/Non-use values of biodiversity
Source: U.S. EPA (1999)		

Economic analyses of air pollution control have paid less attention to ecological benefits than direct benefits to human health. There is a complex and nonlinear relationship between ecosystem damages and air pollution, and many impacts are difficult to measure.

The most important ecological benefits of air pollution abatement include:

- Eutrophication* of estuaries associated with atmospheric nitrogen deposition
- Reduced tree growth associated with ozone exposure
- Acidification of freshwater bodies associated with atmospheric nitrogen and sulfur deposition
- Accumulation of toxics in freshwater bodies associated with atmospheric toxics deposition
- Aesthetic damages to forests associated with ozone and airborne toxics

* Eutrophication is a condition in an aquatic ecosystem where high nutrient concentrations stimulate blooms of algae. Increased eutrophication from nutrient enrichment due to human activities is one of the leading problems facing some estuaries in the Mid-Atlantic region.

Not all ecological benefits are quantifiable or can be monetized. For that reason, in valuation studies attention is often restricted to ecological impacts associated with service flows to humans, rather than broad structural changes to ecosystems. The main criteria for including service flows in valuation studies are that they must be identifiable, quantifiable and monetizable. Table 4.3 summarizes several service flows that satisfy these criteria.

Table 4.3 Candidate Endpoints for Quantitative Assessment			
Ecological Effect	Endpoint	Dose-Response Functions	Economic Model
Acidification	1. Forest Aesthetics	1. Not required	1. Site-specific
	2. Recreational Fishing	2. Multiple available	2. Site-specific
	3. Existence Value of Biodiversity	3. Multiple Available	3. Site-specific
Eutrophication	1. Recreational Fishing		1. Site-specific
	2. Existence Value of Biodiversity		2. None available
Toxics Deposition	1. Forest Aesthetics	1. Multiple available	1. Site-specific
	2. Recreational Fishing	2. Multiple available	2. Site-specific
	3. Existence Value of Biodiversity	3. Multiple available	3. None available
	4. Hunting and Wildlife Aesthetics	4. Multiple Available	4. Site-specific
Multiple Pollutant Stress	1. Ecosystem aesthetics and ecosystem existence value	1. None available	1. None available
Source: U.S. EPA (1999)			

There are four primary methods to monetize non-health-related benefits:

1. Hedonic property value methods
2. Travel cost methods (TCM)
3. Expressed preference methods/Contingent valuation
4. Market models

The use of hedonic property value methods and contingent valuation to monetize ecological benefits is analogous to the valuation of human health benefits. TCM exploits observed differences between travel distance and environmental quality of recreation site to estimate the monetary value of each site characteristic. Market Models study the impact of changes in ecological services on both producers and consumers of market goods that rely on these services.

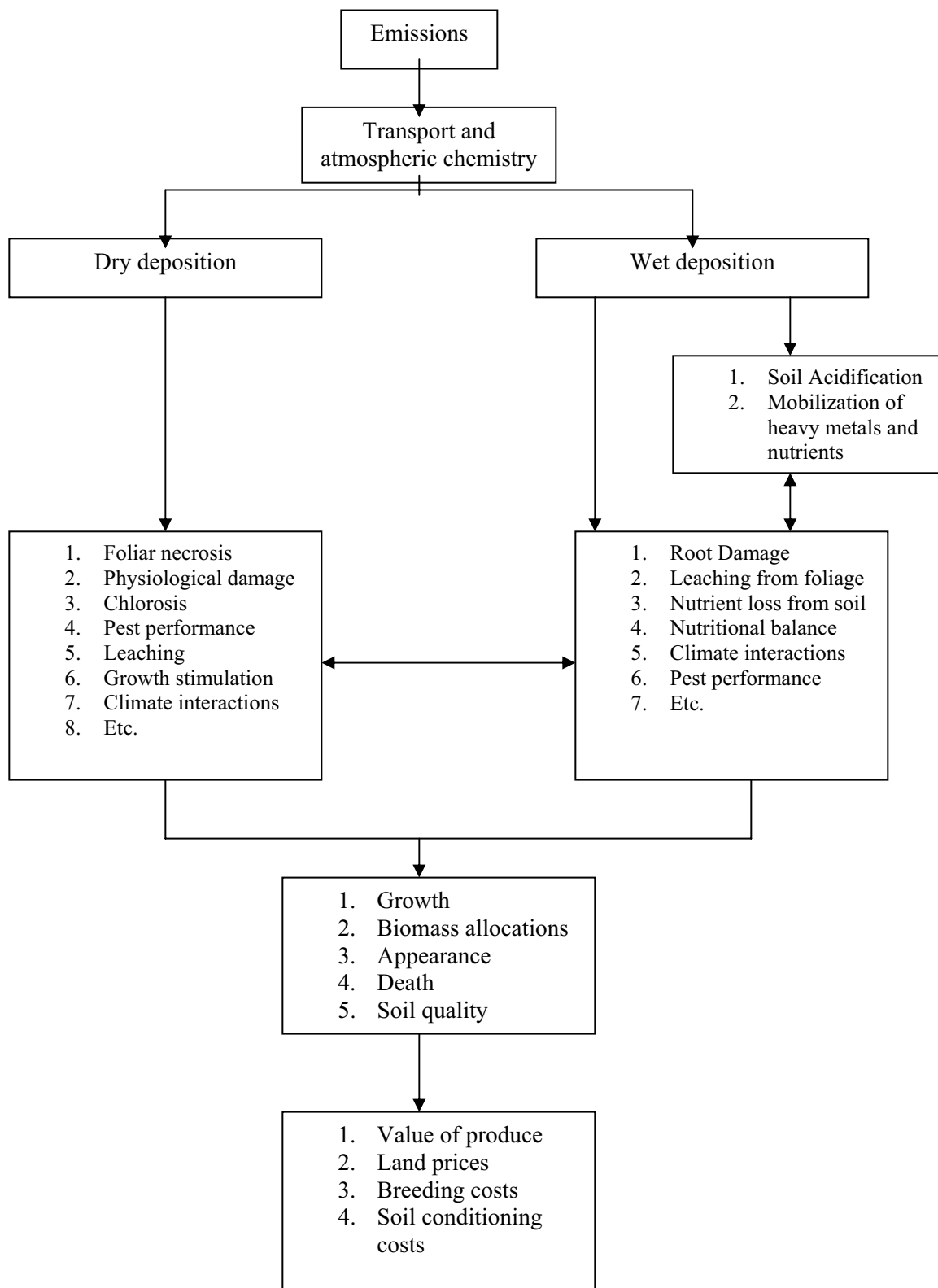
4.1. AGRICULTURAL PRODUCTIVITY BENEFITS

Air pollution has a negative impact on agricultural productivity. Research in the area has focused primarily on the economic impacts of tropospheric ozone, acidic deposition, and global climate change on commercial crops. Economic assessments of the impact of air pollution on crop losses are sometimes associated with forestry impacts. However, as Spash (1997) points out, forestry is a multi-product production system, in which the economic valuation of the impacts pertains to a much wider set of issues, including biodiversity changes, reduced aesthetics and recreation services. Some of these impacts have only non-use values, and therefore forestry damages are poorly represented in market-related production models that are commonly used to estimate agricultural productivity damages. Therefore, in what follows we review only the methodology commonly used to estimate agricultural crop losses due to air pollution.

Pollutants that have been found to have a negative impact on crop yields are ozone (O_3) and its precursor pollutants (mostly NO_x), and sulfur dioxide (SO_2). Chart 4.1 illustrates the various impact pathways of air pollution on agricultural crops. Of all the pollutants, the most extensive research in recent years has been conducted on tropospheric ozone. Ashmore (1991) concludes that although gaseous pollutants other than ozone (namely, SO_2 and NO_2) may be locally important at high concentrations, they have little economic impact on a national scale. Only minor damage to plants has been attributed to gaseous pollutants other than ozone and to sulfate and acid deposition. The National Crop Loss Assessment Network (NCLAN) program found statistically significant response to SO_2 only in soybeans and tomatoes. Herrick and Kulp (1987) report a negligible impact of SO_2 and NO_2 within the National Acid Precipitation Assessment Program (NAPAP). Ashmore (1991) finds that barley, clover, and lucerne are especially sensitive to SO_2 , but these are minor crops associated with small potential economic benefits of pollution control.

Acidic deposition is an indirect impact pathway between air pollution and crop yields. While gaseous pollutants affect crops directly by foliage or above ground exposure, acidic deposition causes changes in soil chemistry due to additions of sulfur and nitrogen. The various positive (e.g. passive fertilization) and negative effects of acidic deposition could potentially neutralize each other, although the final outcome is highly dependent on edaphic conditions and crop cultivar (Spash, 1997).

Chart 4.1. Impact pathway illustrating the effects of air pollution on agricultural crops
Source: Holland et al. (2002)



For example, Adams et al. (1986) included fertilization effects of nitrate deposition, and compared them with additional expense for lime used to reduce acidity. The economic analysis of a 50% increase in acidic deposition resulted in a net benefit.

Corn and soybean appear to be the most sensitive crops to acidic deposition. In addition, the available research seems to suggest that most commercial crop yields are relatively insensitive to acidic deposition on its own (Segerson, 1991). Spash (1997) concludes that crop damages from SO₂, NO₂ and acid deposition combined are 5-10% of the crop damages of ozone pollution.

Segerson (1987) has identified a number of factors that distinguish the effects of acidic deposition from those of ozone:

- Acidic deposition affects a wide range of non-market goods while ozone affects mostly commercial crops
- Acidic deposition is a dynamic pollutant that accumulates over time, while ozone is periodic. Therefore an economic analysis of ozone exposure may be based on short-term studies, while acidic deposition should be based upon assessing accumulated impacts over time.
- The impacts of acidic deposition are surrounded by a greater degree of uncertainty than those of ozone.
- Ozone pollution is a more localized problem than acidic deposition.

Ozone has been observed to cause significant damages in terms of crop yield losses at current ambient levels. Furthermore, the increased frequency and duration of hot dry weather implied by global warming will increase the concentration of tropospheric ozone from available precursors. The Table 4.1.1 below illustrates the damages to crops from ozone exposure

Growth	Development	Yield	Quality
Rate	Fruit set & development	Number	Appearance: size, shape, and color
Pattern	Branching	Mass	Storage life
	Flowering		Texture/cooking quality, Nutrient content, Viability of seeds

Source: Jacobson (1982) p. 296, Table 14.1.

Although ozone-induced quality degradation may be a significant part of total economic damages, research has almost exclusively focused on estimating changes in output resulting from air pollution. There is currently insufficient information available as to the importance of crop quality response (Spash, 1997).

The majority of economic assessments of ozone damage to crops have been at the regional level. Published studies have concentrated on two main regions of the U.S., namely, the Corn Belt (Illinois, Indiana, Iowa, Ohio and Missouri) and California.

4.1.1 QUANTIFYING AGRICULTURAL PRODUCTIVITY BENEFITS

Dose-response functions describe the relationship between ambient ozone concentrations and changes in crop yields. There are three main approaches for deriving dose-response relationships:

1. Foliar injury models
2. Secondary response data
3. Experimentation.

Foliar injury models assess visible injury symptoms, quality changes, and growth responses to air pollution, and they often require making subjective judgments by the researcher. These were the primary methods used in the early literature. Another method of dose-response function estimation is the use cross-sectional analysis of crop yield data via regression techniques. Data on outdoor pollutant concentrations, actual crop yields, and other environmental factors are need for the dose-response relationship estimation. Examples of the use of secondary response data include Leung et al. (1982) and Rowe and Chestnut (1985). Experimental approaches to study the effects of ozone on crops include the use of greenhouses, field chambers (open-top and closed-top), unenclosed plots, and the pollution gradient approach.

Using experimental methods, the National Crop Loss Assessment Network (NCLAN) developed concentration-response relationships linking ground-level ozone to leaf damage and reduced seed size in an effort to determine the effect of ozone on crop yield. Estimated minimum and maximum dose-response functions for six major crops are summarized in the table below.

Table 4.1.2 Ozone Exposure Response Functions for Selected Crops					
Ozone Index	Quantity	Crop	Dose-response Function	Median Experimental Duration (Days)	Median Duration (Months)
SUM06	Max	Cotton	$1 - \exp\left(-(\text{index}/78)^{1.311}\right)$	119	4
SUM06	Min	Cotton	$1 - \exp\left(-(\text{index}/116.8)^{1.523}\right)$	119	4
SUM06	Max	Field Corn	$1 - \exp\left(-(\text{index}/92.4)^{2.816}\right)$	83	3
SUM06	Min	Field Corn	$1 - \exp\left(-(\text{index}/94.2)^{4.307}\right)$	83	3
SUM06	Max	Grain Sorghum	$1 - \exp\left(-(\text{index}/177.8)^{2.329}\right)$	85	3
SUM06	Min	Grain Sorghum	$1 - \exp\left(-(\text{index}/177.8)^{2.329}\right)$	85	3
SUM06	Max	Peanut	$1 - \exp\left(-(\text{index}/99.8)^{2.219}\right)$	112	4
SUM06	Min	Peanut	$1 - \exp\left(-(\text{index}/99.8)^{2.219}\right)$	112	4
SUM06	Max	Soybean	$1 - \exp(-\text{index}/131.4)$	104	3
SUM06	Min	Soybean	$1 - \exp\left(-(\text{index}/299.7)^{1.547}\right)$	104	3
SUM06	Max	Winter Wheat	$1 - \exp(-\text{index}/27.2)$	58	2
SUM06	Min	Winter Wheat	$1 - \exp\left(-(\text{index}/72.1)^{2.353}\right)$	58	2
Source: USEPA (1999) Originally from Lee and Hogsett (1996)					

The commonly assumed form of the dose-response function is the Weibull function, which has the following functional form:

$$Q = \mu \cdot e^{-\left(\frac{O_3}{r}\right)^\lambda}$$

where Q is the observed yield, μ is the hypothetical yield at zero ozone exposure, O_3 is the ozone concentration (ppb) and r is the ozone concentration when the yield is 0.37 μ and λ is a shape parameter. This functional form is often used in empirical analysis partly because of its biologic plausibility.

4.1.2 VALUATION OF AGRICULTURAL PRODUCTIVITY BENEFITS

The traditional approach to valuing crop losses due to air pollution was to calculate monetary equivalents of the approximated losses by multiplying decreased yields by the current market price to give a producer benefit estimate equal to total revenue. It has been shown that this method is likely to overestimate the gain to producers from ozone reductions. In addition, traditional methods often ignored farmers' reactions in terms of changing the input mix and cross-crop substitution.

In more recent empirical work agricultural production models have been used to estimate the economic costs associated with yield losses due to air pollution. These models estimate the social benefit from reduced ozone damages. The social benefit from a reduction in the ozone air pollution is the change in total surplus minus the change in deficiency/transfer payments. The social benefit, or total surplus, consists of producer surplus (the total difference between the market price and the willingness-to-supply on each unit sold) and the consumer surplus (the total difference between the market price and the willingness-to-pay on each unit bought). On one hand, a reduction in crop damage reduces costs, and hence increases the supply of the crop. This contributes to an increase in the producer surplus, *ceteris paribus*. On the other hand, altered levels of ozone pollution may affect the attributes of a crop, thus changing the consumers' willingness to pay, and consequently change the consumer surplus. The agricultural production model calculates the competitive equilibrium that maximizes the total surplus subject to resource constraints.

Various agricultural production models have been used in economic assessments of ozone damage. Murphy et al. (1999) use the AOM8 estimate the welfare changes due to ozone air pollution the markets for eight major crops (corn, soybeans, wheat, alfalfa hay, cotton, grain sorghum, rice, barley). The effects of a reduction in ozone concentrations are modeled as a shift in the production function—at lower ozone levels, more output is obtained from a given set of inputs. AOM8 is a modified version of the agricultural production model used in Howitt (1991). In response to output and input price changes, AOM 8 accounts for endogenous price effects and substitution of cropping activities.

Howitt et al. (1984) studied 13 crops using the California Agriculture Resources Model (CARM) to calculate consumer and producer surpluses. CARM is a quadratic programming model allowed for constrained cross-crop substitution.

The U.S. EPA (1999) study used the Agricultural Simulation Model (AGSIM©, Taylor et al., 1993). The model is able to simulate the markets (equilibrium prices and quantities) for ozone-sensitive crops. AGSIM© is an econometric-simulation model that is based on a large set of statistically estimated demand and supply equations for agricultural commodities produced in the United States.

The use of an agricultural production model requires one to specify an agricultural production function. For example, Murphy et al. (1999) use a Cobb-Douglass production

function, where land, water, nitrogen, and pesticides are the inputs. This specification assumes that a given change in ozone concentrations causes the same percentage change in output for any combination of the inputs. To estimate the changes in producer and consumer surplus, the shift in the production function is estimated based on dose-response functions for individual crops. In Murphy et al. (1999) the relationship between production levels under the baseline and the policy scenario are given by the following formula:

$$Q_i^P = (1 + \frac{QGAIN\%_i}{100})Q_i^B$$

Where Q_i^S and Q_i^B are production levels of crop i under the policy scenario and the baseline scenario, respectively, and $QGAIN\%_i$ is the percentage change in the yield of crop i resulting from a reductions in ambient ozone concentrations induced by the policy. The percentage change in the output, $QGAIN\%_i$, is estimated using dose-response functions. Each ozone reduction scenario results in a unique set of optimal input quantities, equilibrium output prices and quantities, and welfare measures (total, consumer, and producer surplus).

4.2 RECREATIONAL AND COMMERCIAL FISHING BENEFITS

Recreational and commercial fishing benefits form a subgroup of ecological benefits of air pollution. The theoretical basis for valuing ecological benefits in general, and recreational and commercial fishing benefits in particular, is that the natural environment provides us with services that we value (Freeman, 1997). There are no suitable methods to comprehensively value many of these service flows (e.g. breathable air, livable climate). Therefore in valuation studies, we are limited to valuing services flows that are either material inputs into our economy, or provide amenities associated with marketed services (e.g. recreation).

Three types of pollution are associated with commercial and recreational fishing: acidification, nitrogen eutrophication, and toxics deposition. Acidification, or acid deposition, is probably the best-studied effect of air pollution on ecosystems. The main cause of acidification is acidic precipitation in the form of sulfuric acid (H_2SO_4) and nitric acid (HNO_3). These acids are formed from sulfur dioxide (SO_2) and nitrogen oxides (NO_x) found in the atmosphere. Electric power plants are among the primary point sources of SO_2 . On the other hand, a large share of NO_x is from non-point sources (transportation), and therefore anthropogenic NO_x is more dispersed in the atmosphere compared with anthropogenic SO_2 . Deposition occurs via three main pathways: (1) wet deposition, where the pollutant is dissolved in precipitation, (2) dry deposition, which is a direct form of deposition of gases and particles to any surface, and (3) cloud-water deposition, when cloud or water droplets are intercepted by vegetation. Since most of the precipitation falls on the terrestrial part of the catchments, soil properties are generally

strongly associated with water quality. Consequently, acidification resulting from acid deposition usually occurs in areas with acidic soils. Throughout the world freshwater acidification is the most serious in eastern parts of the United States and Canada, and in Europe, particularly in Scandinavia. Reductions of anthropogenic SO₂ and NO_x emissions in Europe in recent years have resulted in an improvement in acidified water bodies, however, the same trend has not been observed in the United States (Stoddard et al., 1999). It is believed that it may take ecosystems several decades to recover from the impact of acidification even after emissions have been cut.

Eutrophication is the result nitrogen deposition leading to excessive nitrogen enrichment of aquatic ecosystems, and it may adversely affect the biogeochemical cycles of watersheds. Atmospheric nitrogen is deposited into water bodies through dry and wet deposition. Excessive eutrophication can lead, for example, to massive algae booms, which in turn reduces the oxygen levels and leads to habitat loss. It is estimates that 86% of the East Coast Estuaries are susceptible to eutrophication (U.S. EPA, 1997).

Toxics deposition involves hazardous air pollutants (HAPs), as defined by the Clean Air Act: mercury, polychlorinated biphenyls (PCBs), chlordane, dioxins, and dichlorodiphenyltrichloro-ethane (DDT). When considering air pollution from power plants, the most important of these pollutants is mercury. Much of mercury found in ecosystems comes from natural sources. Mercury accumulates in fish, birds, and mammals, and it may be dangerous to humans when the concentration exceeds a critical level. Mercury-based statewide fish consumption advisories are fairly common in the United States.

Table 4.2.1 below summarizes the main recreational and commercial fishing impacts associated with acidification, eutrophication, and toxics deposition.

Table 4.2.1 Recreational and commercial fishing can be associated with the following ecological impacts of air pollution:		
Pollutant Class	Ecosystem Effect	Service Flow
Acidification (H ₂ SO ₄ , HNO ₃)	Freshwater acidification resulting in fish (and other aquatic organism) decline	Recreational Fishing
Nitrogen Eutrophication (NO _x)	Freshwater acidification resulting in fish (and other aquatic organism) decline	Recreational Fishing
Toxics Deposition (Mercury, Dioxin)	Aquatic bioaccumulation of mercury and dioxin	Recreational and Commercial Fishing
Source: U.S. EPA (1999)		

4.2.1 Quantifying Recreational and Commercial Fishing Benefits

Following emissions modeling, and transport and deposition modeling, the next step in the process of quantifying the benefits to recreational fisheries is the use of an exposure

model. Unfortunately, comprehensive models to quantify the impacts of acidification, eutrophication, and toxics deposition are currently not available.

To quantify the impact of acid deposition on fisheries, US EPA (1999) uses a region-specific model to quantify the effects of acidification on freshwater fish populations: Model of Acidification of Groundwater Catchments – MAGIC (Cosby et. al. (1985a,b). MAGIC is calibrated to the watershed of an individual lake or stream and then used to simulate the response of that system to changes in atmospheric deposition. The model simulates the effects of acid deposition on both soils and surface waters. The simulation typically involves seasonal or annual time steps and is implemented on decadal or centennial time scales.

Quantifying Eutrophication

When atmospheric nitrogen is deposited in estuaries, it can lead to eutrophication. Estimation of a dose-response relationship between nitrogen loading and water quality changes is complicated because of the dynamic nature of ecosystems. Most likely, these dose-response functions are nonlinear with a threshold. Unfortunately, universally transferable dose-response functions for quantifying the effects of eutrophication have not yet been developed. USEPA (1999) study quantified deposition-related nitrogen loadings for three estuaries (Chesapeake Bay, Long Island Sound, Tampa Bay) using GIS-related methods. Data on nitrogen deposition, together with information on abatement options to reduce excess nutrient loads, was used for valuation purposes. In addition, USEPA (1999) used specific biophysical indicators of estuarine health to quantify the benefits. This approach is useful when there is a direct link between the biophysical indicator and the ecological service flows. USEPA (1999) used the properties of the seagrass bed, which provides habitat for variety organisms, and have been shown to decline with increased nitrogen deposition, as an indicator.

Quantifying Toxics Deposition

Most damages to ecosystems are caused by five hazardous air pollutants (HAP): mercury, PCBs, dioxins, DDT, and chlordane. The mechanism of ecosystem responses to toxic contamination is poorly understood. Furthermore, service flow impacts of ecosystem damages are difficult to observe because HAPs persist in aquatic ecosystems for a long time. A comprehensive quantitative analysis with the available models and data is currently not possible.

4.2.2 Valuation of Recreational and Commercial Fishing Benefits

Unlike commercial fishing, recreational fishing is to a large part a non-market activity. Although most states charge a license fee for recreational fishing, the license fee itself is believed not to reflect the true willingness-to-pay (WTP) for recreational fishing. Marginal WTP for recreational fishing is a function of catch attributes (e.g. number and the average size of fish caught) and other determinants. Environmental factors indirectly affect WTP for recreational fishing by affecting the catch attributes. The total value of recreational fishing to the angler can be measured by the consumer surplus, which is the difference between WTP and the actual amount they pay or the cost they incur for the recreational fishing day. Consumer surplus is measured by the area below the demand curve and above the price or the cost of a recreational fishing day.

There is an extensive literature on valuation of fishing opportunities by anglers. In the valuation literature, two primary methods have been used most often to deduce the value of recreational fishing: travel cost method (TCM), and contingent valuation method (CVM). TCM uses observed travel costs to visit a fishing site and per-capita visitation rates to deduce the demand for recreational fishing. On the other hand, CVM is questionnaire-based method, where anglers are asked hypothetical question about how much they would be willing to pay for a day of fishing.

TCM cannot be used to measure willingness-to-accept (WTA) some degradation in an environmental amenity (i.e. compensation demanded for an environmental damage). Hence, TCM cannot be used to estimate the value of loss of fishing opportunity due to air pollution. Furthermore, the use of CVM in general has generated controversy in the valuation literature. CVM is subject to an inherent bias due to its hypothetical nature. Study participants are often subjected to an unfamiliar market context, or they may not be fully aware of the characteristics of the good in question, or their own budget constraints. Some critics of CVM have pointed out that estimates of WTP obtained using CVM may not reflect the true WTP for the non-market good, but they rather reflect the WTP pay for moral satisfaction. The answers from CVM studies may be biased because of passive-use motives, such as the “warm glow” effect (Andreoni, 1989). Individuals’ responses to WTP questions serve the same function as charitable contributions, and people are assumed to get a “warm glow” from giving. Some economists do not fully recognize “warm glow” as an economic value.

In contrast to many earlier studies utilizing TCM or CVM, Snyder et al. (2003) use a revealed-preference approach to estimate the value of recreational fishing. Their method to estimate WTP for a recreational fishing day is based on observed fishing license sales. Unlike CVM and TCM that require extensive micro-level data, Snyder et al. (2003) are able to use aggregate, state-level data for their estimation. The following simple model describes the methodology used in Snyder et al (2003). A representative consumer’s utility depends on the number of recreational fishing days, X , the fishing license L , and all other goods denoted by a composite good Z .

$$U = U(X, L, Z)$$

Because X and L are complementary, we can write the following:

$$\frac{\partial U(X, 0, Z)}{\partial X} = 0$$

$$\frac{\partial U(0, L, Z)}{\partial X} = 0$$

That is, the marginal utility of a fishing license or a recreational fishing day alone is zero. The demand for annual fishing licenses can be estimated from the observable data on license sales. By measuring the appropriate area under the demand curve, one can estimate the average benefits per license, and consequently, the value of a recreational fishing day.

Several problems may arise when using this method. One is that the assumed complementarity between the fishing license and recreational fishing day holds only in the absence of illegal fishing. A comprehensive economic model of consumer behavior would account for illegal fishing by including the “price” of illegal fishing in the estimation process. As Snyder et al. (2003) note, in most cases, fines are set by courts, and therefore omitting fines should not bias the analysis, as there is no apparent correlation between license fees and fines. Another potential problem associated with the method of Snyder et al. (2003) is price endogeneity, that is, that causality runs not only from price to quantity, but vice versa. To address possible price endogeneity, the authors use the instrumental variable (IV) estimation technique, using instruments that are exogenous to the demand for fishing licenses. The set of instruments includes variables that are indicative bureaucratic and political proclivities of states, such as the size of the government, and the degree of regulation and taxation.

Snyder et al. (2003) obtain estimates of the value of a recreational fishing day for 48 U.S. states. Table 4.2.2 summarizes the results for Mid-Atlantic states that are most likely to be affected by the New Jersey RPS.

Table 4.2.2 Estimates of the value of a freshwater recreational fishing day for selected states Snyder et al. (2003) (2000\$'s)										
State	Linear IV	Semi-log IV	Log-log IV cutoff \$32.8	Log-log IV cutoff \$100	Log-log IV cutoff \$200	Log-log IV cutoff \$500	90% confidence interval based on semi-log		90% confidence interval based on log-log (\$200 cutoff)	
New Jersey	1.33	1.29	0.23	0.37	0.50	0.74	0.25	6.65	0.00	3.21
New York	1.45	1.29	0.32	0.51	0.69	1.03	0.30	5.59	0.01	3.49
Pennsylvania	2.21	3.00	2.01	3.26	4.40	6.56	0.57	15.87	2.06	7.43
Maryland	1.66	1.38	0.34	0.55	0.74	1.11	0.35	5.44	0.01	3.63
Delaware	1.08	0.79	0.18	0.30	0.40	0.60	0.23	2.74	0.00	2.13
Notes: i. Estimates are based on instrumental variables regressions on data from 1975-1989 ii. Models of demand for three functional forms are estimated: linear, multiplicative (log-log) and semi-log. For each functional form, two specifications are reported: one includes all relevant substitutes of resident annual licenses, and the other includes only the price of short-term Type 1 licenses and dummy variables for each year. iii. Cutoff values are used as upper limits to integrate the demand function.										

A comparison of Snyder et al. (2003) estimates with the values from other studies reveals that there is a considerable geographic variation in the estimated value of recreational fishing. Moreover, the estimates are significantly lower than those reported in other studies employing TCM or CVM. The differences could be due to methodological differences, as well as, to the elimination of the biases in TCM and CVM.

Table 4.2.3 Comparison of Snyder et al. (2003) estimates with other studies					
Study	Estimation Method	Study Location	Type of Fishing	Valuation (2000 \$'s)	Valuation Snyder et al. (2003) ⁱ (2000 \$'s)
King and Hof (1985)	TCM	Alabama	Trout	24.57	5.86 (5.22, 32.14)
Miller and Hay (1980)	TCM	Arizona	All	73.14	2.90 (1.09, 5.14)
Walsh et. al. (1980)	CVM	Colorado	Cold water	22.01	10.40 (10.01, 30.38)
Ziemer et.al. (1980)	TCM	Georgia	Warm water	27.65	2.92 (2.80, 5.59)
Miller and Hay (1980)	TCM	Idaho	All	56.42	11.12 (10.80, 25.18)
Loomis and Sorg (1986)	TCM	Idaho	Cold water Warm water	45.59 47.04	11.12 (10.80, 25.18)
Miller and Hay (1980)	TCM	Maine	All	48.06	4.43 (2.73, 6.88)
Miller and Hay (1980)	TCM	Minnesota	All	60.60	18.80 (13.34, 305.88)
Haas & Weithman (1982)	TCM	Missouri	Trout	27.97	4.86 (4.62, 8.06)
Dutta (1984)	TCM	Ohio	Cold water	8.73	3.51 (2.60, 5.24)
Brown and Shalloof (1984)	TCM	Oregon	Salmon Steelhead	36.47 49.72	12.97 (12.29, 26.94)
Kealy and Bishop (1986)	TCM	Wisconsin	All	51.60	11.55 (10.37, 52.00)
Notes: i. Snyder et al. (2003) estimates are based on the log-log instrumental variable specification with \$200 cutoff; 90% confidence interval in brackets					

Another limitation of many early studies is that they do not include a direct measure of water quality. A notable exception is Montgomery and Needelman (1997) that consider the special case of toxic contamination of fisheries. Toxic contamination is a special case of pollution, because contaminants in fish become dangerous to humans eating fish before they result in a decline in fish population. In addition, through health advisories the public is better informed about incidences of toxic contamination than other forms of pollution (e.g. acidification or eutrophication).

Montgomery and Needelman (1997) employ the Random Utility Model, which is a site-choice model. A brief description of the RUM model follows. Utility associated with recreational fishing for individual *i* in fishing site *j* is given by the following equation:

$$U_{ij}^F = V_{ij}^F + \varepsilon_{ij}^F$$

where V_{ij}^F is the observable portion of the utility function, and ε_{ij}^F is a random error. Standard RUM models assume that V_{ij}^F is a linear function of income, M_i , cost of visiting the site, P_{ij} , and a vector of site characteristics, X_j .

$$V_{ij}^F = \beta_M(M_i - P_{ij}) + \beta_X X_j$$

where β_M and β_X are parameters to be estimated. Using the parameter estimates we can estimate the inclusive value, I_i , which is the maximum utility that an individual taking a trip receives.

$$I_i = \ln \sum_{j=1}^J e^{\mu \beta_X X_{ij} - \mu \beta_M P_{ij}}$$

The utility of not fishing is represented by the following equations.

$$U_i^N = V_i^N + \varepsilon_i^N$$

$$V_i^N = \beta_M M_i + \beta_Z Z_i$$

Given the set of individual attributes, Z_i , the probability that an individual will go fishing on a given day is given by the following formula:

$$\Pr_i(fish = 1) = \frac{e^{\frac{1}{\mu} I_i}}{e^{\frac{1}{\mu} I_i} + e^{\beta_Z Z_i}}$$

The economic value for an individual of improved water quality (compensating variation) can be calculated as follows:

$$4CV_{it} = \frac{\ln(e^{\frac{1}{\mu} \hat{I}_i^2} + e^{\hat{\beta}_Z Z_i}) - \ln(e^{\frac{1}{\mu} \hat{I}_i^1} + e^{\hat{\beta}_Z Z_i})}{\hat{\beta}_M}$$

Table 4.2.3 Valuation studies of recreational fishing using RUM			
Study	Location	Study Period	Type of Fishing
Morey et al. (2001)	Maine rivers	1988	Salmon fishing
Ahn et al. (2000)	North Carolina mountain streams	1996	Trout fishing
Jakus et al. (1998)	Tennessee reservoirs	1997	Recreational fishing
Lupi and Hoehn (1998)	Great Lakes, Michigan	1994	Trout and salmon fishing
Parsons et al. (1998)	Maine lakes and rivers	1989	Recreational fishing
Pendleton and Mendelsohn (1998)	New England lakes	1989	Recreational fishing
Schumann (1998)	North Carolina	1987-1990	Ocean Fishing
Train (1998)	Montana rivers and lakes	1987-1990	Recreational fishing
Greene et al. (1997)	Tampa Bay, Florida	1991-1992	Recreational fishing
Hoehn et al. (1997)	Michigan lakes and rivers	1994-1995	Recreational fishing
Montgomery and Needelman (1997)	New York lakes	1989	Recreational Fishing
Feather et al. (1995)	Minnesota lakes	1989	Recreational fishing
McConnell and Strand (1994)	Mid- to South-Atlantic	1987-1988	Recreational ocean fishing

4.3 BIODIVERSITY BENEFITS

Biodiversity is a valuable environmental amenity. A number of human actions, including anthropogenic air pollution, have led to a dramatic decline in biodiversity across the globe (Pimm et al., 2001). Biodiversity refers not just to an accumulation of species in a given area, but it also incorporates the ecological and evolutionary interactions among them (Armsworth et al., 2004).

The first step in estimating biodiversity benefits is defining biodiversity. Biodiversity encompasses four levels as it is summarized in the table below:

Table 4.3.1	
Type of Biodiversity	Physical Expression
Genetic	Genes, nucleotides, chromosomes, individuals
Species	Kingdom, phyla, families, genera, subspecies, populations
Ecosystem	Bioregions, landscapes, habitats
Functional	Ecosystem, functional, robustness ecosystem resilience services goods
Source: Turner et al. (1999)	

Genetic biodiversity is the most basic level, and it refers to the information represented in the DNA of living organisms. Species-level biodiversity refers to the variety of species in a given area. Because only a small fraction of the estimated 5-30 million species currently living on the earth (Wilson, 1988) have been identified and described, empirical estimates of the species-level biodiversity are often surrounded by a great degree of uncertainty. Community-level biodiversity is important, because it is believed that species-level diversity enhances the productivity and stability of ecosystems (Nunes and van den Bergh., 2001, Odum, 1950). However, recent studies suggest that no pattern or determinate relationship may exist between species-level diversity and stability of ecosystems (Nunes and van den Bergh. 2001, Johnson et al. 1996). Functional diversity, or the ecosystem's functional robustness, refers to the ability of the ecosystem to absorb external shocks. Unfortunately, the ecosystem's functional diversity is still poorly understood (Nunes and van den Bergh, 2001).

Human threats to biodiversity include activities causing habitat loss (conversion, degradation or fragmentation) and climate change, harvesting, as well as the introduction of exotic species that by becoming dominant competitors or effective predators may drive many native species to extinction.

The electric utility sector contributes to habitat degradation (acidic deposition and eutrophication) by emitting nitrogen oxides (NO_x) and sulfur oxides (SO_x), as well as to climate change by emitting greenhouse gases into the atmosphere.

Empirical estimates of Morse et al. (1995) and Field et al. (1999) of the impact of climate change on biodiversity illustrate the magnitude of threats to biodiversity: 4 °F average temperature increase can reduce the number of all species in California by 5%-10%.

4.3.1 *Quantifying Biodiversity Benefits*

The following discussion on quantifying biodiversity benefits is based on Armsworth et al. (2004). The traditional approach to measuring biodiversity has focuses on species-level biodiversity, which can be measured in two ways:

- *Richness* – number of species in a given area
- *Evenness* – how well distributed abundance or biomass is among species within a community

Evenness is the greatest when species are equally abundant. For example an area that has a total population of 100 of 10 different species, each comprising of 10 individuals, is more diverse than a community of 1 species with 91 individuals and 9 other species with one individual each.

A diversity index is an overall measure of diversity that usually combines aspects of richness and evenness. One of the most commonly used diversity index is the Shannon-Weiner index (H') defined below.

$$H' = -\sum_{i=1}^S p_i \ln(p_i)$$

where the summation is over all species (i.e., S is the total number of species at site), and p_i is the relative abundance of species i (i.e., p_i is the number of individuals of species divided by the total number of individuals of all species at the site). H' is high if there are many species, or if evenness is high.

Example Calculation of H':

Suppose we study a 1-acre area in a forest and have counted 240 redbud trees, 120 post oak trees, 40 black hickory tree, and 320 red oak trees. Species richness calculations are summarized in the table below:

Species (<i>i</i>)	N_i	p_i	$\ln(p_i)$	$p_i \ln(p_i)$
Redbud trees (1)	240	0.333	-1.100	-0.366
Post oak tree (2)	120	0.167	-1.792	-0.298
Black hickory tree (3)	40	0.056	-2.882	-0.161
Red oak tree (4)	320	0.444	-0.811	-0.361
Total	720	1.000		-1.186

Estimating relative abundances for all species can be time-consuming and difficult, therefore species-level richness is often used as a proxy for species-level biodiversity. In contrast to the Shannon-Weiner index this simplification places relatively large weight on rare species. Typically, measures of species-level diversity are not applied to all species at site, but rather to a particular taxonomic group, such as, mammals, insects, or plants (Armsworth et al. 2004)

The choice of the spatial scale is also important, because richness increases with the size of the area. The appropriate scale is typically an economically meaningful scale (e.g., individual land parcel) or an ecologically meaningful scale (e.g., habitat zone).

Once the spatial scale has been chosen, there are three aspects of biodiversity to consider (Whittaker 1972, Schluter and Ricklefs, 1993):

- α -diversity – the “local” diversity within each site
- β -diversity – the change in species composition from one site to another
- γ -diversity – the “total” diversity measured over the entire suite of sites being considered

When ecological data are not available, ecological or biological production functions may be used to approximate the changes to biodiversity. One of the most robust and useful ecological patterns that researchers have observed is the species-area relationship. This relationship is often approximated by a power-law formula:

$$S = cA^z$$

where c and z are positive constants.

This relationship describes a static pattern of biodiversity, and it is not informative about the composition of the local community. The simplest representation of the dynamics of a closed community takes the form:

$$\dot{N}_i = N_i f^i(N_1, \dots, N_S)$$

for $i = 1, \dots, S$

S = number of interacting species, f^i = per capita growth rate of each species; f^i can depend on all species' densities. The relevant partial densities indicate whether the interaction between any two species in the community is cooperative ($f_{N_j}^i > 0, f_{N_i}^j > 0$), predatory or parasitic ($f_{N_j}^i < 0, f_{N_i}^j > 0$) or competitive ($f_{N_j}^i < 0, f_{N_i}^j < 0$). A few functional forms of ecological production functions have been reported in the literature (e.g. Roughgarden, 1997).

4.3.2 Valuation of Biodiversity Benefits

The monetization of biodiversity benefits requires one to assess what it is about biodiversity that consumers value. In general, consumers' benefit can be divided into *use value* (direct such as tourism or indirect such as pollination) or *non-use* (intrinsic or existence) *value*. Direct benefits to consumers arise in two important ways:

- *Service flows* – Ecosystems provide valuable services to society, such as water purification in natural watersheds, prevention of soil erosion and carbon sequestration by standing forests, and recreational services such as ecotourism and birdwatching. The service flow approach to valuation dictates that investments in preserving or restoring biodiversity need to deliver a competitive return relative to other investment opportunities within the economy, and hence it does not necessarily maximize α , β or γ diversity (Armsworth, 2004).
- *Bet hedging* – Conserving biodiversity provides society with a bet-hedge against unforeseen circumstances. For example if society were overly reliant on monocultured ecosystems, it would be vulnerable to catastrophic losses in service provision due to disease outbreaks. Hence there is a bet-hedging benefit to conserving γ diversity.

Nonuse or *existence value* of biodiversity refers to the utility consumers derive from knowing that certain species (still) exist.

Nunes and van den Bergh (2001) critically evaluate a number of biodiversity valuation studies at each level of biodiversity value contained in Table 4.3.1. The authors conclude that available economic valuation estimates should be regarded as providing a very

incomplete perspective on the value of biodiversity changes, and they provide at best the lower bounds on that value. The tables below summarize the results from the valuation studies that have been performed in North America. Table 4.3.2 lists the results from contingent valuation studies estimating the individual WTP to avoid the loss of a particular species. The main shortcoming of these single-species valuation studies is that they do not account for species substitution and complementary effects.

Table 4.3.2 Biodiversity value estimates from single-species valuation studies		
Author(s)	Study	Mean WTP estimates (per household per year)
Stevens et al. (1997)	Restoration of the Atlantic salmon in one river, Massachusetts	\$14.38-21.40
Loomis and Larson (1994)	Conservation of the Gray Whale, US	\$16–18
Loomis and Helfand (1993)	Conservation of various single species, US	From \$13 for the Sea Turtle to \$25 for the Bald Eagle
Van Kooten (1993)	Conservation of waterfowl habitat in Canada's wetlands region	\$50–60 (per acre)
Bower and Stoll (1988)	Conservation of the Whooping Crane	\$21–141
Boyle and Bishop (1987)	Two endangered species in Wisconsin: the Bald Eagle and the Striped Shiner	From \$5 for the Striped Shiner to \$28 for the Bald Eagle
Brookshire et al. (1983)	Grizzly Bear and Bighorn Sheep in Wyoming	From \$10 for the Grizzly Bear to \$16 for the Bighorn Sheep
Source: Nunes and van den Bergh (2001)		

Multiple-species valuation studies account for all related species, and the resulting estimates are in general higher than those of single-species studies. Multiple-species valuation studies are summarized in Table 4.3.3 below.

Table 4.3.3 Biodiversity value estimates from multiple-species valuation studies		
Author(s)	Study	Mean WTP Estimates (per household per year)
Desvousges et al. (1993)	Conservation of the migratory Waterfowl in the Central Flyway	\$59–71
Whitehead (1993)	Conservation program for coastal nongame wildlife	\$15
Duffield and Patterson (1992)	Conservation of fisheries in Montana Rivers	\$2–4 (for residents) \$12–17 (for non residents)
Halstead et al. (1992)	Preservation of the Bald Eagle, Coyote and Wild Turkey in New England	\$15
Samples and Hollyer (1989)	Preservation of the Monk Seal and Humpback Whale	\$9.6–13.8
Hageman (1985)	Preservation of threatened and endangered species populations in the US	\$17.73–23.95
Source: Nunes and van den Bergh (2001)		

Table 4.3.4 summarizes estimates from valuation studies that link the value of biodiversity to the value of natural habitat conservation.

Table 4.3.4 Biodiversity value estimates from natural habitat valuation studies		
Author(s)	Study	Mean WTP estimates (per household)
Richer (1995)	Desert protection in California, US	\$101
Kealy and Turner (1993)	Preservation of the aquatic system in the Adirondack Region, US	\$12–18

Table 4.3.4 Biodiversity value estimates from natural habitat valuation studies (continued)		
Author(s)	Study	Mean WTP estimates (per household)
Hoehn and Loomis (1993)	Enhancing wetlands and habitat in San Joaquin valley in California, US	\$96–184 (single program)
Diamond et al. (1993)	Protection of wilderness areas in Colorado, Idaho, Montana, and Wyoming, US	\$29–66
Silberman et al. (1992)	Protection of beach ecosystems, New Jersey, US	\$9.26–15.1
Boyle (1990)	Preservation of the Illinois Beach State Nature Reserve, US	\$37–41
Loomis (1989)	Preservation of the Mono Lake, California, US	\$4–11
Smith and Desvousges (1986)	Preservation of water quality in the Monongahela River Basin, US	\$21–58 (for users) \$14–53 (for nonusers)
Mitchell and Carson (1984)	Preservation of water quality for all rivers and lakes, US	\$242
Walsh et al. (1984)	Protection of wilderness areas in Colorado, US	\$32
Source: Nunes and van den Bergh (2001)		

Table 4.3.5 summarizes estimates from valuation studies that link the value of biodiversity to the value of natural areas with high tourism and outdoor recreation demand.

Table 4.3.5 Biodiversity value estimates from ecosystem functions and services valuation studies			
Author(s)	Study	Measurement method	Estimates
Laughland et al. (1996)	Value of a water supply in Milesburg, Pennsylvania, US	Averting behavior	\$14 and 36 per household
Table 4.3.5 Biodiversity value estimates from ecosystem functions and services valuation studies (continued)			
Author(s)	Study	Measurement method	Estimates
Abdalla et al. (1992)	Groundwater ecosystem in Perkasio, Pennsylvania, US	Averting behavior	\$61 313–131 334
McClelland et al. (1992)	Protection of groundwater program, US	Contingent valuation	\$7–22
Torell et al. (1990)	Water in-storage on the high plains aquifer, US	Production function	\$9.5–1.09 per acre-foot
Ribaudo (1989a,b)	Water quality benefits in ten regions in US	Averting behavior	\$4.4 billion
Huszar (1989)	Value of wind erosion costs to households in New Mexico, US	Replacement costs	\$454 million per year
Holmes (1988)	Value of the impact of water turbidity due to soil erosion on the water treatment, US	Replacement costs	\$35–661 million annually
Walker and Young (1986)	Value of soil erosion on (loss) agriculture revenue in the Palouse region, US	Production function	\$4 and 6 per acre
Source: Nunes and van den Bergh (2001)			

Finally, Table 4.3.6 provides ranges of estimates for the various levels of biodiversity value derived from the studies reviewed by Nunes and van den Bergh (2001).

Table 4.3.6 Summary of biodiversity values by biodiversity level			
Biodiversity level	Biodiversity value type	Value ranges	Method(s) selected
Genetic and species diversity	Bioprospecting	From \$175 000 to \$3.2 million	Market contracts
	Single species	From \$5 to 126	Contingent valuation
	Multiple species	From \$18 to 194	Contingent valuation
Ecosystems and natural habitat diversity	Terrestrial habitat (non-use value)	From \$27 to 101	Contingent valuation
	Coastal habitat (non-use value)	From \$9 to 51	Contingent valuation
	Wetland habitat (non-use value)	From \$8 to 96	Contingent valuation
	Natural areas habitat (recreation)	From \$23 per trip to 23 million per year	Travel cost, tourism revenues
Ecosystems and functional diversity	Wetland life-support	From \$0.4 to 1.2 million	Replacement costs
	Soil and wind erosion protection	Up to \$454 million per year	Replacement costs, hedonic price, production function
	Water quality	From \$35 to 661 million per year	Replacement costs, averting expenditure
Source: Nunes and van den Bergh (2001)			

4.4 FORESTRY BENEFITS

Air pollution has been recognized as a potential problem for forests for a long time. Sulfur dioxide, fluorides, heavy metals and ozone pose the greatest threat to forest ecosystems. In the past, sulfur emissions, that cause acid rain, were the primary concern, but in recent decades massive efforts to reduce this pollutant have been largely successful. Today, in terms forestry impacts ozone may be the pollutant associated with the greatest potential benefits.

Scientific evidence suggests that elevated tropospheric ozone levels disrupt vegetation growth, and interfere with the respiratory function of plants carried out by photosynthesis even at concentrations below current air quality standards (Wang et al., 1986; Reich and

Amudson, 1985). Sometimes ozone injury to plants has observable effects such as yellowing or stippling of leaves, but negative impacts of ozone often occur without accompanying visible symptoms.

Ambient ozone enters the plant through pores in the leaf or needle called stomata, where most of the plant's metabolic and respiratory activity occurs. Once ozone enters these stomata, it initiates a chain reaction that destroys or damages plant proteins and enzymes, as well as the fatty chemicals that help form cell membranes. Plants continue to suffer damage long after the ozone exposure episode is over. Furthermore some researchers have suggested that there are synergies between ozone and acid deposition (Hewitt, 1990).

Ozone damage to forests is a common problem in many parts of the eastern U.S. Particularly sensitive species to ozone are the poplars (*Populus* spp.), white pine (*Pinus strobus*), and the oak family (*Quercus* spp.).

Another serious threat to forest ecosystems is acid deposition in the form of nitrogen acids due to nitrogen oxides emissions. Aluminum is naturally present in forest soils in the form of chemical compounds that are harmless to living organisms. Nitrogen acids cause ions of aluminum to become mobile in soil, and its toxic form, aluminum is taken up into the tree's roots. This may result in reduced root growth, which reduces the tree's ability to take up water and withstand drought. Excess nitrogen is also absorbed directly from the air through the leaves during fog and low clouds. If ozone is present in sufficient concentrations, exposure to this oxidant can damage the leaves, damaging respiration processes of the organism.

Given the evidence on damage to forests and the size of the forest cover, forestry benefits seem to play an important role in total benefits due to reduced air pollution in the northeastern United States. According to Mid-Atlantic Integrated Assessment (MAIA) – multi-agency effort headed by the USEPA to assess the health and sustainability of ecosystems – forests cover 61% of the total land area in the MAIA region⁹. Ninety-five percent of the region's forests are classified as timberland. The vast majority (79%) of timberlands is owned by nonindustrial private landowners, while the forest industry owns approximately 7%. Hardwood forests dominate the MAIA region. For the region as a whole, oak /hickory is the predominant forest type. Other dominant forest types in the region are northern hardwoods, loblolly/shortleaf pine, and oak/pine.

4.4.1 Quantifying Forestry Benefits

Due to the different life cycles involved, the assessment of forest damage is substantially more difficult than that of agricultural crop damage. On one hand trees live for a long time, which makes the study of pollution impacts much more difficult. On the other hand,

⁹ The MAIA study region includes Delaware, Maryland, Pennsylvania, Virginia, and West Virginia, and parts of New Jersey, New York, and North Carolina.

unlike agricultural soils, which are effectively managed through annual cycles, forest soils are much less disturbed which leads to an accumulation of acidification impacts.

USEPA (1999) uses the PnET-II model to estimate the impacts of tropospheric ozone on commercial timber growth. The PnET model simulates the cycles of carbon, water, and nitrogen through forest ecosystems. Model inputs of monthly weather data and nitrogen inputs are used to predict photosynthesis, evapotranspiration, and nitrogen cycling on a monthly time-step for several forest types.

4.4.2 Valuation of Forestry Benefits

The valuation techniques of forestry benefits can be grouped into three categories:

1. direct market prices
2. indirect market prices
3. hypothetical values

Methods using direct market prices are based on actual prices, and consequently they do not reflect some benefits (e.g. preservation of biodiversity) that the market participants did not take into account in their decision-making. Methods utilizing indirect market prices include hedonic property values, the travel costs, opportunity costs, surrogate prices and replacement costs (Cavatassi, 2004). The opportunity cost method uses the market price of the best alternative forgone to provide a lower bound on forestry benefits. Surrogate prices methods use the market price of a close substitute as a proxy for the benefits. A surrogate market approach is used by methods using hypothetical values. Two methods that belong to this category are the contingent valuation method, and conjoint analysis.

As described in the previous sections, contingent valuation method uses surveys that ask hypothetical questions to estimate economic values. Conjoint analysis estimates values by asking people by asking people questions across a range of features or attributes of a forest^φ. Forestry benefits may be grouped into three categories for valuation purposes: on-site private benefits, on-site public benefits, and global benefits. On-site private benefits include timber productions, agricultural and other agroforestry products, and non-timber forest products (e.g. mushrooms, medicinal plants, honey, fruits, nuts, etc.), and recreation and tourism. On-site public benefits include watershed protection, agricultural productivity enhancement, nutrient cycling, microclimate regulation, and aesthetic, cultural, and spiritual values. Global benefits include carbon sequestration, and biodiversity conservation.

Values derived for some of these benefits are not transferable, and therefore most valuation studies restrict attention to on-site benefits such as timber production. These benefits are usually estimated using market models. For example, USEPA (1999) used

^φ Cavatassi (2004)

the USDA Forest Service Timber Assessment Market Model to estimate market changes that result from reduced timber growth.

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5. INDIRECT BENEFITS TO HUMANS THROUGH NONLIVING SYSTEMS

There are two types of indirect benefits through nonliving systems that were studied the most: avoided materials damages, and improved visibility. The steps of estimating these benefits are summarized in the sections that follow.

5.1 AVOIDED MATERIALS DAMAGES

Anthropogenic sulfur and nitrogen pollutants are believed to have caused vast damage to buildings, structures, as well as the cultural heritage in the past century. Much of the damage occurred in Europe and North America, but with growing car traffic, and very high concentrations of sulfur dioxide in many cities of China, India, and Latin America material damages due to air pollution continue to remain a significant problem.

Objects most susceptible to air pollution are the ones with long lives, particularly buildings. Other objects, such as cars, may be damaged by air pollution, but these damages tend to be less important, because they are usually replaced before the damage could become significant.

Pollutants that contribute to degradation of buildings are particles (particularly soon) causing soiling, and sulfur dioxide (SO₂) contributing to corrosion and erosion caused by acid rain.

Principal effects associated with air pollution are:

- loss of mechanical strength
- leakage,
- failure of protective coatings,
- loss of details in carvings,
- pipe corrosion.

SO₂ has a strong accelerating effect on the degradation of certain materials by contributing to corrosion by acidic deposition. Atmospheric corrosion is influenced by climatic patterns such as relative humidity, temperature and precipitation, and it tends to be a local problem, because the damage often occurs near the source of emission. On the other hand, indirect effects of SO₂ emissions caused the acidification of soil and water bodies, tend to be a regional problem due to the long-range transport of air pollutants.

Air pollution damages materials such as zinc, copper, stone, as well as organic materials. In case of zinc and copper, the dissolution of protective corrosion products leads to increased deterioration rates. Calcareous stones, such as limestone or marble, are very susceptible to acid deposition by sulfur dioxide through transformation of the original calcium carbonate to gypsum and calcium sulfate. Degradation of organic materials, such

as rubber tires and paints, are usually associated with ozone in conjunction with temperature and solar radiation.

5.1.1 Quantifying materials damages

The ideal approach of quantifying materials damages is analogous to the approach used to quantify health endpoints. One would start by estimating the change in pollutant concentrations caused by a policy-induced reduction in emissions. The second step would involve the use of dose-response or concentration-response (CR) functions that relate the physical damage to ambient pollutant concentrations. And the last step involves attaching monetary values to damages.

A valid CR function provides a mathematical relationship between properties of the environment and some index of materials, such as loss of stock thickness. Some early attempts aimed at deriving such CR functions focused on the relationship between ambient pollutant concentrations and corrosion rates. As Lipfert (1996) points out, this approach neglects the important variable of delivery of the reactant to the surface. A full understanding of the process requires a separation of pollutant delivery process from the subsequent chemical reactions. The appropriate technique to estimate a CR function is that of a multiple regression, in which some index of corrosion is the independent variable and the various environmental factors are the independent variables.

Perhaps the most difficult element of economic assessment of materials damages (or benefits from reduced air pollution) is the problem of estimating stocks of buildings at risk (Lipfert and Daum, 1992). One problem is that there is considerable heterogeneity in the use of housing materials across country. While residential housing materials tend to follow regional patterns, commercial and industrial buildings tend to be more uniform. Another important problem since atmospheric corrosion has been present in many parts of the country for a long time, people may have substituted away from the more sensitive building materials toward less sensitive ones. The greatest difficulty lies in distinguishing chemical characteristics of exposed surfaces within each building type and category. Lipfert (1996) suggests that there is a need for a probabilistic, as opposed to deterministic, approach to assessment. There are many relevant but disparate databases on building stocks, but no effort has been made yet to synthesize that information (Lipfert, 1996).

As an alternative to the above bottom-up approach, Rabl (1999) estimates damages to buildings in France by working with aggregate data on observed frequencies of cleaning and repair activities. The result is a “combined concentration-response function”. The main variables in the CR function are income and the ambient concentration of particulate matter.

Rabl (1999) considers two types of damage caused by air pollution:

- Corrosion or erosion of coatings and construction materials

- Soiling

Corrosion and erosion are primarily due to acid deposition. A large number of studies analyzed the effects of air pollution and corrosion and erosion. Dose-response functions have been estimated for several building materials (e.g., Kucera, 1990; Haneef et al., 1992; Butlin et al., 1992, Lipfert, 1987). There are relatively few studies on soiling due to air pollution, and consequently few dose-response functions are available (Hamilton and Mansfield, 1992).

5.1.2 Valuation of materials damages

Given a valid dose response function, using the bottom-up approach one would estimate the repair cost due to air pollution as follows:

Total Repair Cost = Sum Surface Area * Repair Frequency * Repair Cost

Using a bottom-up approach the following are the steps in valuations:

- Division into pollution strata
- Materials inventory and inspection of physical damage
- Damage functions
- Estimated change in service life
- Maintenance/ Replacement cost
- Estimated economic damage

The main drawback of the bottom-approach is the need for very detailed data on building inventories. As an alternative to the bottom-up approach, Rabl (1999) uses a linear regression of renovation expenditures against income, PM_{13} , and SO_2 . The best regression model was the following:

$$R = \beta_0 + \beta_1 Income + \beta_2 PM_{13}$$

where R and Income are measured in monetary units per person per year, while, PM_{13} is the measure of particulate matter concentrations in $\mu g/m^3$, and β_0 , β_1 and β_2 are parameters to be estimated from the data. The above equation is what Rabl (1999) calls a combined or aggregate concentration-response function. The change in repair costs in response to a change in pollutant concentrations is then given by the following expression:

$$\frac{\partial R}{\partial PM_{13}} = \beta_2 Income$$

Neither approach to valuation may be used to assess damages to the cultural heritage. Cultural heritage encompasses both outdoor buildings and sculptures and treasured objects kept indoors, stored in museums and archives. The most appropriate valuation method for assessment is contingent valuation. These valuation studies tend to be case specific and generally not transferable.

5.2 VISIBILITY BENEFITS

Reduced visibility due to anthropogenic air pollution affects some of the country's most scenic areas. US EPA estimates that in national parks in the eastern United States, average visual range has decreased from 90 miles to 15-25 miles. In the West, visual range has decreased from 140 miles to 35-90 miles. The main cause of visibility impairment is haze. Under stagnant air mass conditions, aerosols can be trapped and produce a visibility condition usually referred to as layered haze. Some light is absorbed by particles while other light may be scattered away before it reaches the observer. The introduction of particulate matter and certain gases into the atmosphere therefore reduces visibility.

From a technical point of view, visibility is a complex and difficult concept to define. Visibility includes psychophysical processes and concurrent value judgments of visual impacts, as well as the physical interaction of light with particles in the atmosphere. Therefore it is important to understand the psychological process involved in viewing a scenic resource, and to be able to establish a link between the physical and psychological processes.

5.2.1 *Quantifying visibility benefits*

Quantifying visibility requires developing links between visibility and particles that scatter and absorb light. Visibility, in the most general sense, reduces to understanding the effect that various types of aerosol and lighting conditions have on the appearance of landscape features. Measuring visibility by a single index is, however, not possible because visibility cannot be defined by a single parameter (Malm, 1999). Many visibility indices have been proposed, however the most simple and direct way of communicating reduced visibility is through a photograph. In fact, many contingent valuation (CV) studies of visibility present the subjects with photographs of scenic areas with varying levels of visibility. The reason photographs communicate visibility changes so well is that the human eye works much like a camera. The human eye detects relative differences in brightness rather than the overall brightness level, that is to say, the eye measures contrast between adjacent objects.

Because the human eye function like a camera, a photograph captures visibility changes, as humans perceive it. However, it is difficult to extract quantitative information from photographs, and therefore direct measure of fundamental optical measures of the atmosphere have been developed. The most common measures are atmospheric extinction and scattering.

The scattering coefficient is a measure of the ability of particles to scatter photons out of a beam of light, while the absorption coefficient is a measure of how many photons are absorbed. Both coefficients are expressed as a number proportional to the amount of photons scattered or absorbed per distance. The sum of scattering and absorption is referred to as extinction or attenuation.

5.2.2 Valuation of visibility benefits

The most commonly used methods for visibility valuation are hedonic property values and contingent valuation. Hedonic methods are based on revealed preference of consumers, because they link nonmarket valuation to a traded commodity. Hedonic property value studies estimate the marginal WTP function on the basis of an estimated relationship between housing prices and housing attributes (including air quality). There are several factors that affect the relationship between property values and air quality. These include adverse health effects, reduced visibility or soiling due to air pollution. Hedonic methods cannot be used to estimate separately. Disaggregation of overall impacts requires making subjective judgments by the researcher. Nevertheless, the results of hedonic property value studies confirm the hypothesis that air quality has a significant impact on property prices. Kenneth and Greenstone (1998) estimate that the Clean Air Act induced nationwide monetized benefits were \$80 billion (in 1982-84 dollars) in the 1970's, and \$50 billion during the 1980s. Delucchi, Murphy and McCubbin (2002) estimate monetized costs of total suspended particle pollution in 1990 at \$52-\$88 billion in (1990 dollars). Some studies (e.g., Brookshire et al., 1979, 1982; Loehman et al., 1994; McLelland et al., 1991) attempted to disaggregate property value impacts into health, visibility, soiling, and other impacts. They find that visibility impacts are the second most important, after health effects, representing 19-34% of total monetized benefits.

Burtraw et al. (1997) present the results of an integrated assessment of the benefits and costs of the Title IV of the 1990 Clean Air Act Amendments initiated reductions in emissions of sulfur dioxide and nitrogen oxides. They use the Tracking and Analysis Framework (TAF) developed for the National Acid Precipitation Assessment Program (NAPAP). Although uncertainties surround their estimates, the findings suggest that the benefits of the program substantially outweigh its costs. Two types of visibility effects are examined: recreational visibility at two national parks (Grand Canyon and Shenandoah), and residential visibility in five metropolitan areas (Albany, NY, Atlantic City, NJ, Charlottesville, VA, Knoxville, TN, and Washington, DC). The results, summarized in Table 5.2.1 below, are most usefully considered on a per capita basis.

Table 5.2.1 Per Capita Benefits in 2010 for Affected Population

Effect	Benefits per Capita (1990\$)
Morbidity	3.50
Mortality	59.29
Aquatic	0.62
Recreational Visibility	3.34
Residential Visibility	5.81
Costs	5.30
Source: Burtraw et al. (1997), Table 2, pp. 13	

These visibility estimates illustrate their potential magnitude, but it should be noted that they are based on relatively small number of studies available in the literature, and also the geographical scope of the project is rather limited. Burtraw et al. (1997) explain the relatively large magnitude of visibility benefits compared to other types of benefits, namely aquatics, by claiming that willingness to pay depends on the availability of substitutes, and visibility, along with health, has no close substitutes.

Smith and Osborne (1996) perform a meta-analysis of visibility valuation studies to test whether CV estimates of WTP are responsive to the amount, or scope, of the environmental amenity being offered. They consider an internal consistency test for CV-based WTP. Internal consistency tests assess the reliability and validity of CV surveys. One way to evaluate the CV method is to compare willingness to pay WTP functions estimated with CV surveys with the specific, observable properties that economic theory implies WTP should follow. Smith and Osborne (1996) selected five of CV studies that used comparable methods for the meta-analysis. These studies focused on air quality as a key element. Furthermore, in each study air quality is presented in a way that permits computation of the change in visible range. The five selected studies are summarized in Table 5.2.2 below.

Table 5.2.2 Summary of CV studies for visibility at national parks analyzed by Smith and Osborne (1996)				
Authors	Mean and inter-quartile range of WTP (per month in 1990 \$)	Mean change in visibility	Location	Type of survey
Rowe et al. (1980)	\$9.27 (\$6.83, \$10.82)	0.50	Navaho Recreation Area	In-person interviews administered to households in area
MacFarland et al	\$2.75 (\$1.69, \$3.73)	1.18	Grand Canyon and Mesa Verde National Parks	In-person interviews administered to visitors to the area
Schulze et al.	\$8.50 (\$4.42, \$11.67)	0.79	Grand Canyon, Mesa Verde, and Zion National Parks	In-person interviews administered to households in Albuquerque, Los Angeles, Denver, and Chicago
Chestnut and Rowe	\$4.35 (\$3.15, \$5.48)	0.62	Grand Canyon, Yosemite, and Shenandoah National Parks	Mail with telephone households in Arizona, Virginia, California, New York, and Missouri
Balson et al.	\$0.46 (\$0.007, \$0.97)	0.955	Grand Canyon National Park	In-person interviews conducted in St. Louis and San Diego Counties
Source: Smith and Osborne (1996), pp. 291, Table 1				

The findings of Smith and Osborne (1996) support a positive, statistically significant and robust relationship between the WTP estimates and the percentage improvement in visible range. These results suggest that it may be possible to transfer results from a meta-analysis of past CV studies. The crucial issue in benefit transfer is to find a common metric to measure the environmental amenity.

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6. SAMPLE CALCULATIONS

The following examples illustrate the steps and calculations involved in the estimation of non-market benefits of the RPS. The examples below focus on mortality and morbidity benefits only.

Step1: Projecting Concentrations

In this step the change in pollutant concentrations around each power plant is estimated. Instead of using a more sophisticated dispersion model, we used the Gaussian Plume Dispersion Model (GPDM) to project pollutant concentrations. This model predicts an average concentration under steady state conditions. The shape of the plume undergoing dispersion is a function of the wind speed, vertical temperature profile and atmospheric stability. GPDM is widely used to predict concentrations in the atmosphere. There are, however, significant simplifications with this model. The main assumptions of GPDM are:

1. Only steady-state concentrations are estimated.
2. Wind blows in x-direction and is constant in both speed and direction
3. Transport with the mean wind is much greater than turbulent transport in the x-direction.
4. The source emission rate (the rate at which the pollutant is emitted per unit of time) is constant.
5. Diffusion coefficients are constant in both time and space.
6. The source emits chemicals of concern (COC) at a point in space $x=y=0$ and $z=H$, where H is the effective stack height.
7. The COC's are inert (non-decaying and non-reactive).
8. There is no barrier to plume migration.
9. Mass is conserved across the plume cross section.
10. Mass within a plume follows a Gaussian (normal) distribution in both the crosswind (y) and vertical (z) directions.

The GPDM is derived from the advection-diffusion equation. The general equation to calculate the steady state concentration of an air contaminant in the ambient air resulting from a point source is given by:

$$C(y, x, z) = \frac{Q}{2\pi u_y \sigma_z} \exp\left(\frac{-y^2}{2\sigma_y^2}\right) \left\{ \exp\left(\frac{-(z-H)^2}{2\sigma_z^2}\right) + \exp\left(\frac{-(z+H)^2}{2\sigma_z^2}\right) \right\}$$

where

$C(x,y,z)$ = contaminant concentration at the specified coordinate
 x = downwind distance

y = crosswind distance
 z = vertical distance above ground
 Q = contaminant emission rate
 σ_y = lateral dispersion coefficient function
 σ_z = vertical dispersion coefficient function
 u = wind velocity in downwind direction
 H = effective stack height

In the above equation σ_y , the lateral dispersion coefficient function, and σ_z , the vertical dispersion coefficient functions depend on the downwind distance and the atmospheric stability class. The value of these coefficients in meters can be obtained from the equations utilized by the Industrial Source Complex (ISC) Dispersion Model developed by USEPA (1995):

$$\sigma_y = 465.11628 \cdot x \cdot \tan(\text{TH})$$

where

$$\text{TH} = 0.01745 \cdot [c - d \ln(x)]$$

$$\sigma_z = ax^b$$

A simplified version of the above formula to estimate steady state pollutant concentrations is given by

$$C(y, x, z) = \frac{Q}{2\pi\pi u_y \sigma_z} \exp \left\{ -\frac{1}{2} \left(\frac{y^2}{\sigma_y^2} + \frac{(z-H)^2}{\sigma_z^2} \right) \right\}$$

This formula assumes that the pollutant is not reflected from the ground, and therefore it yields lower estimates in general than when one assumes reflection.

EXAMPLE 1: Reduction in excess mortality due to SO₂

Power plant assumptions

We have in mind a coal-burning power plant with the following characteristics:

Capacity = 800MW

Capacity Factor = 0.7[¶]

SO₂ emission factor per output: 10 lbs/MWh

Contaminant emissions rate (g/s): 705

Physical stack height (m): 125

Effective stack height (m): 215.96

Consistent with the followings assumptions:

Stack velocity (m/s): 15

Stack exit diameter (m): 1.5

Stack gas temperature (K): 450

Ambient gas temperature (K): 300

We assume that the RPS results in a 10% reduction in generation by this power plant.

Assumptions about population and health status

Total population: 8.4 million

Total non-accidental deaths: 70,766

Non-accidental mortality per person: 0.00841

Population is assumed to be uniformly distributed in the affected area.

Population density per square kilometer: 402

Meteorological and other assumptions

Wind velocity in downward direction (m/s): 1

Incoming solar radiation during the day: moderate

PASQUILL-GIFFORD category: B

Cloud condition at night: mostly overcast

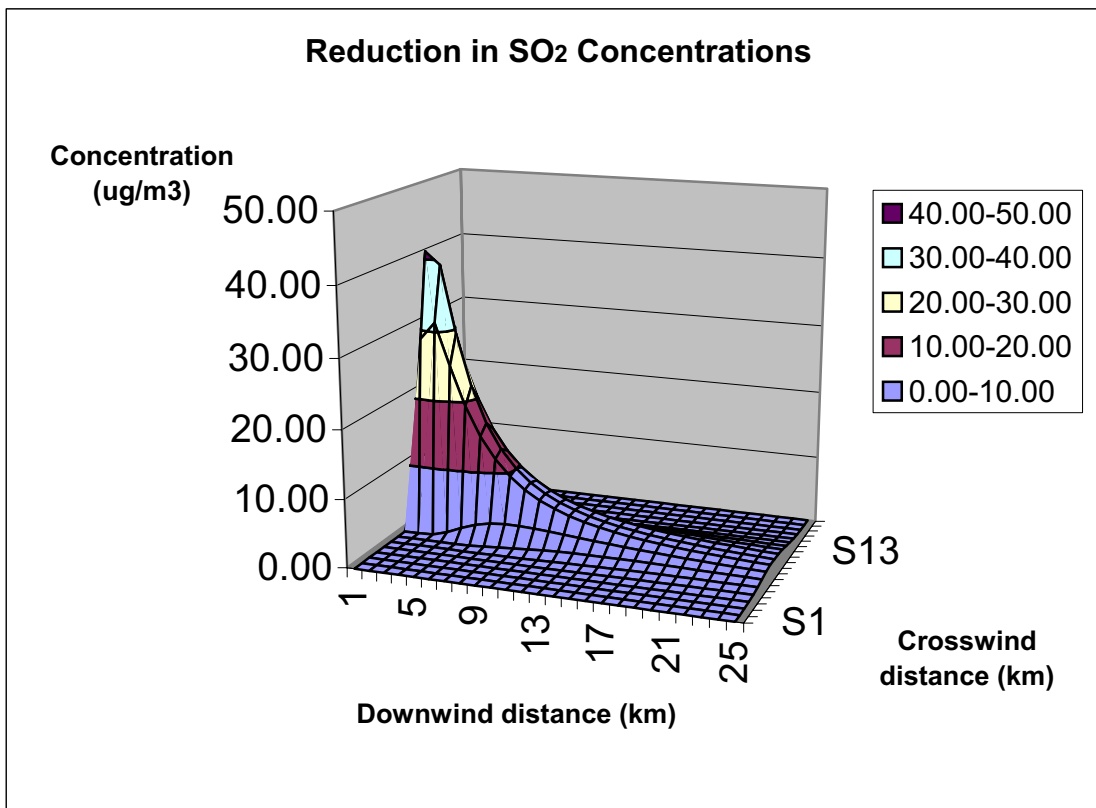
PASQUILL-GIFFORD category: B

Step 2: Quantifying the change in SO₂ mortality

First, using GPDM we calculated SO₂ concentrations for 100-meter grids for 25 km downwind and 9 km crosswind distances. Next, the estimated concentrations were

[¶] Capacity factor is the ratio of the electrical energy produced by a generating unit for the period of time considered to the electrical energy that could have been produced at continuous full power operation during the same period.

averaged at the 1-km grid level. These steps were followed for both the baseline scenario and the policy scenario (10% reduction in generation). The difference in SO₂ concentration between the two scenarios at the 1-km grid level is estimated.



Next concentration response functions are used to calculate the change in non-accidental mortality.

CR-function:

$$\Delta \text{mortality} = - \left(y_1 (e^{\beta \Delta \text{SO}_2} - 1) \right) \cdot \text{pop}$$

- y_1 = non-accidental deaths per person
- β = SO₂ coefficient
- ΔSO_2 = change in SO₂ concentrations (ppb)
- pop = population sample

We used the SO₂ mortality CR function estimated by Touloumi et al. (1996). The 95% confidence interval (CI) for the reduction in non-accidental mortality (statistical deaths) is: (27.5, 72.2)¹ with an estimated value 30.89.

¹ This CI refers only to the uncertainty associated with the CR-function estimation.

Step 3: Monetizing SO₂ mortality benefits

Using the median estimate of Viscusi and Aldy (2003) for the Value of Statistical Life (VSL) of \$7 million, the estimated benefit is \$216.2 million with a 95% confidence interval of (\$192.5m, 505.4m).

EXAMPLE 2: Reduction in excess chronic obstructive pulmonary disease (COPD) hospital admissions due to PM₁₀

This example illustrates the estimation of morbidity benefits associated with improved air quality. The health benefit of interest is avoided hospital admissions for chronic obstructive pulmonary disease (COPD) due to particulate matter 10 µm and less in diameter (PM₁₀). COPD is a group of diseases categorized by ICD-9-CM codes 490-496.

(ICD=International Classification of Diseases)

- 490 - Bronchitis, not specified as acute or chronic
- 491 - Chronic bronchitis
- 492 - Emphysema
- 493 - Asthma
- 494 - Bronchiectasis
- 495 - Extrinsic allergic alveolitis
- 496 - Chronic airway obstruction, not elsewhere classified

The same power plant, population and meteorological assumptions hold as in the previous example. Morbidity benefits were estimated in the following steps.

Step 1: Determining concentration-response (CR) relationships for COPD admissions.

CR-studies usually assume the following log-linear functional form:

$$\Delta \text{COPD hospital admissions} = - \left(y_1 (e^{\beta \text{PM}_{10}} - 1) \right) \cdot \text{pop}$$

- y_1 = baseline COPD admission rate, defined as COPD hospital admissions per person
- β = estimated PM₁₀ coefficient

ΔPM_{10} = change in PM_{10} concentrations ($\mu g/m^3$)
 pop = exposed population per km^2

Three studies have been identified that estimated the concentration-response relationship between ΔPM_{10} concentrations and hospital admissions for COPD: Chen et al. (2004) in Vancouver, Canada, Zanobetti et al. (2000) in Chicago, Cook county, IL, and Moolgavkar (2000) in Los Angeles county, CA. The parameter estimates are summarized in the table below.

Study	Parameter estimate	95% Confidence Interval		Study Population	Studied Health Effect
	β	β_L	β_H		ICD-9
Chen et al. (2004)	0.0152463	0.0061760	0.0243167	Ages 65+	490-492,494,496
Zanobetti et al. (2000)	0.0076035	0.0015873	0.0136196	Ages 65+	490-492, 494-496
Moolgavkar (2000)	0.0016073	0.0010603	0.0021542	Ages 0-19	490-496
	0.0007968	0.0002110	0.0016826	Ages 20-64	490-496
	0.0009877	0.0004969	0.0014785	Ages 65+	490-496

Step 2: Determining Baseline Exposure

We use the Gaussian Plume Dispersion Model to predict PM_{10} concentration changes. Under our meteorological assumptions most of the deposition occurs within 25 km in downwind direction and 10 km in crosswind direction. We assume that population is uniformly distributed in the affected area. We use a population density estimate derived from 2003 population estimate figures by the U.S. Census Bureau: 1164 people per square mile, or 450 people per square kilometer. Consequently, the total population that is potentially exposed to PM pollutant from the power plant is approximately 250,000.

Based on U.S. Census Bureau population estimates, we assume the following population density estimate for the various age groups:

Age Group	Share of Total Population	Population Density per Square Kilometer
0-19	27.2%	122.1
20-64	59.7%	268.3
65+	13.2%	59.4

Step 3: Determining the Number of Baseline Cases for Each Quantifiable Health Effect Number exposed x Baseline exposure x Dose-response relationship.

Because the health effects studied in the three studies are not identical, it was necessary to estimate and make assumptions about the baseline hospital admission rate for each group of health effects (ICD codes). The Healthcare Cost & Utilization Project (HCUP) by the Agency for Healthcare Research and Quality's (AHRQ) (<http://www.ahrq.gov/hcupnet/>), estimated the number of hospital discharges for the various COPD conditions by various characteristics, such as age, sex, income, etc., at the national and regional (but not state) level. Because our CR-functions are estimated for various age groups, it was necessary to estimate the hospital admission rate for each age group. Hospital admissions were estimated for the following age groups from the 2002 national data:

Age Group	Share of Total Population	ICD 490-492, 494,496	ICD 490-492, 494-496	ICD 490-496
0-17	25.3%	0.00006174	0.00006174	0.0020353
18-64	62.3%	0.00112073	0.00112073	0.0021734
65+	12.4%	0.0116307	0.0116307	0.0136903

Step 4: Determining Exposure for each Policy Scenario, Determining the Number of Cases for Each Quantifiable Effect with the Regulation, Determining the Number of Cases Avoided as a Result of Each Regulatory Option

The avoided COPD hospital admissions for the various age groups and CR functions and dispersion models are summarized in the tables below.

Dispersion Model: Gaussian Plume Dispersion without Reflection					
Estimated Reduction in COPD Hospital Admissions	95% Confidence Interval		CR-Function	Population	Location
6.28	2.40	10.66	Chen et al. (2004)	Ages 65+	Vancouver, Canada
2.98	0.60	5.55	Zanobetti et al. (2000)	Ages 65+ (Medicare admissions)	Chicago, Cook County, IL
0.44	0.22	0.66	Moolgavkar (2000)	Ages 65+	Los Angeles County
0.25	0.07	0.54	Moolgavkar (2000)	Ages 20-64	Los Angeles County
0.22	0.14	0.29	Moolgavkar (2000)	Ages 0-19	Los Angeles County
Dispersion Model: Gaussian Plume Dispersion with Reflection					
Estimated Reduction in COPD Hospital Admissions	95% Confidence Interval		CR-Function	Population	Location
15.81	5.68	28.88	Chen et al. (2004)	Ages 65+	Vancouver, Canada

7.12	1.38	13.81	Zanobetti et al. (2000)	Ages 65+	Chicago, Cook County, IL
1.00	0.50	1.51	Moolgavkar (2000)	Ages 65+	Los Angeles County
0.58	0.15	1.24	Moolgavkar (2000)	Ages 20-64	Los Angeles County
0.50	0.33	0.68	Moolgavkar (2000)	Ages 0-19	Los Angeles County

Step 5: Monetizing the health benefits

Benefits were monetized using the cost-of-illness approach. Using HCUP estimates of mean cost of hospital admissions, weighted cost was estimated for each age group and each groups of ICD codes. The following value were used to monetize benefits:

Age Group	ICD-9 Codes	Average Cost (\$2002)
Ages 65+	490-496	15,537.05
Ages 65+	490-492, 494-496	13,908.82
Ages 65+	490-492, 494, 496	13,886.41
Ages 19-64	490-496	12,421.90
Ages 0-18	490-496	7,511.36

Monetized benefits for the two dispersion models are summarized in the tables below.

Dispersion without reflection	Population	Avoided COPD Admissions (annual)	Monetary Value (\$2002)	95% Confidence interval	
Chen et al. (2004)	65+	6.3	87,186.3	33,319.7	147,969.3
Zanobetti et al. (2000)	65+ Medicare patients	3.0	41,455.3	8,340.3	77,177.5
Moolgavkar (2000)	All ages	0.9	11,583.5	5,319.3	19,087.0

Dispersion with reflection	Population	Avoided COPD Admissions (annual)	Monetary Value (\$2002)	95% Confidence interval	
Chen et al. (2004)	65+	15.8	219,559.5	78,847.9	401,042.0
Zanobetti et al. (2000)	65+ Medicare patients	7.1	98,996.9	19,198.5	192,047.0
Moolgavkar (2000)	All ages	2.1	26,578.3	12,173.5	43,946.7

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Edward J.
Bloustein School
of Planning and Public Policy

Rutgers, The State University of New Jersey
33 Livingston Avenue
New Brunswick, New Jersey 08901-1958